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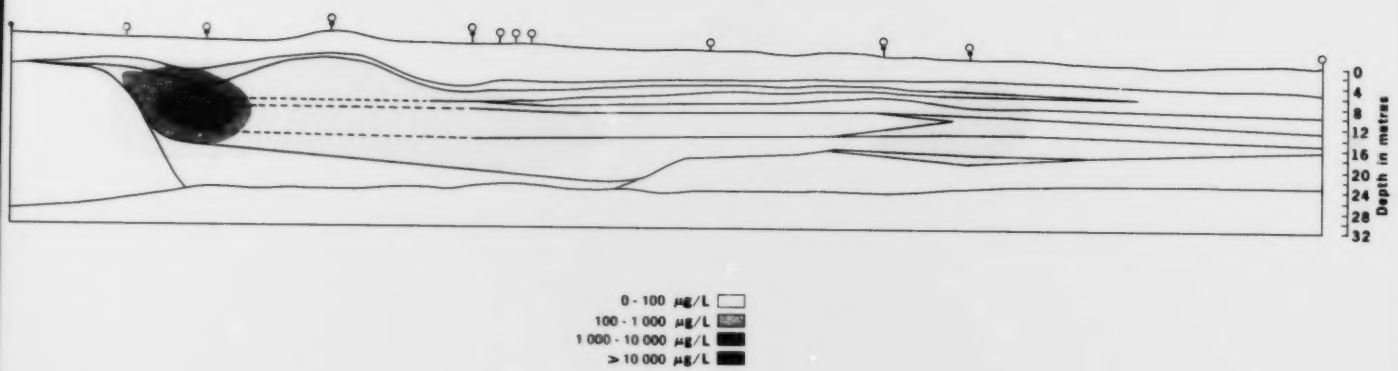
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Depth in metres
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1 000 - 10 000 µg/L
> 10 000 µg/L

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Environment Canada Environnement Canada

Contaminant Hydrogeology of Toxic Organic Chemicals at a Disposal Site, Gloucester, Ontario

1. Chemical Concepts and Site Assessment

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Abstract

Between 1969 and 1980 the federal government disposed of hazardous wastes, principally organic solvents from its laboratories in Ottawa, at the nearby Gloucester Landfill. During 1981, field evidence indicated serious organic pollution emanating from the Landfill in general and from the federal government's Special Waste Compound in particular.

Detailed hydrostratigraphic studies at the site by Gartner, Lee Associates Ltd. in 1978, Geologic Testing Consultants Ltd. in 1982, and the National Hydrology Research Institute during 1981-1982 have resulted in the development of a coherent picture of the outwash aquifer within which the Special Waste plume is migrating. Classical hydrogeological tests were used to assess the hydraulic properties of the aquifer materials; the outwash aquifer exhibits an average hydraulic conductivity of 1.1×10^{-4} m/s. Ground-water velocities were determined by a modified borehole-dilution device which gave an average value of 5.5 cm/day in the outwash aquifer; velocities determined from hydraulic conductivities and from the distribution of bomb tritium in ground water were almost identical.

Patterns of organic pollution emanating from beneath the Special Waste Compound suggest that chromatographic dispersion (i.e. solute transport plus sorption) of the organic contaminants is the principal controlling mechanism and that the octanol-water partition coefficient (K_{ow}) concisely describes this mechanism. Retardation factors derived from plume patterns vary from 1.6 for 1,4-dioxane (log K_{ow} = -0.27) to 23 for carbon tetrachloride (log K_{ow} = 2.83). The outwash aquifer materials contain from 0.2 to 1.0 percent by weight organic carbon, and data suggest that it is less lipophilic than that studied by Schwarzenbach and Westall in Switzerland. Furthermore, it appears that chlorinated ethanes are being transformed in the contaminated ground waters.

Résumé

Entre 1969 et 1980, le gouvernement fédéral a déposé des déchets dangereux, principalement des solvants organiques provenant de ses laboratoires d'Ottawa, à la décharge de Gloucester située non loin de la ville. En 1981, on a décelé des signes d'une pollution organique importante provenant de l'ensemble de la décharge et tout particulièrement de l'aire réservée aux déchets spéciaux du gouvernement fédéral.

Des études hydrostratigraphiques poussées réalisées par Gartner, Lee Associates Ltd. en 1978, par Geologic Testing Consultants en 1982 et par l'Institut national de recherche en hydrologie de 1981 à 1983 ont permis d'établir les caractéristiques de l'aquifère dans lequel migre le panache provenant de l'aire des déchets spéciaux. Des tests hydrologiques classiques ont été effectués pour déterminer les propriétés hydrauliques de l'aquifère. Celui-ci a une conductivité hydraulique moyenne de 1.1×10^{-4} m/s. La vitesse des eaux souterraines, déterminée au moyen d'un dispositif modifié de dilution en trou foré, est en moyenne de 5.5 cm/jour dans l'aquifère, et les vitesses déterminées à partir de la conductivité hydraulique et de la distribution du tritium dans l'eau souterraine étaient presque identiques.

Les caractéristiques de la pollution organique provenant de l'aire des déchets spéciaux semblent indiquer que la dispersion chromatographique (transport des solutés et absorption) des contaminants organiques est le principal mécanisme régulateur et que le coefficient de partage octanol-eau (K_{ow}) décrit bien ce mécanisme. Les facteurs de ralentissement calculés à partir des caractéristiques du panache varient de 1.6 pour le dioxanne-1,4 (log K_{ow} = -0.27) à 23 pour le tétrachlorure de carbone (log K_{ow} = 2.83). L'aquifère contient de 0.2 % à 1.0 % en poids de carbone organique, et les données recueillies indiquent qu'il est moins lipophile que l'aquifère étudié par Schwarzenbach et Westall en Suisse. De plus, il semble que les éthanes chlorés soient transformés dans les eaux souterraines contaminées.

Introduction

TOXIC ORGANIC CHEMICALS AND CANADIAN GROUND WATERS

Approximately 80% of all ground water pumped in Canada for domestic, commercial, agricultural or industrial purposes (or about 20% of total Canadian water use) is withdrawn from shallow sand and gravel aquifers. These are particularly important as regional ground-water supplies in southern Ontario and are locally important in many other areas of Canada (Fig. 1). Many of these aquifers could or have become contaminated because they lack the natural protection afforded them by overlying silt or clay beds.

In southern Ontario alone approximately 67 million litres of liquid industrial wastes was disposed of in 16 landfills during a 12-month period in 1980-81; many of these landfills are in sands and gravels which form good aquifers (e.g. Tiny Township near Midland). On the Prairies, the Regina glaciofluvial sand aquifer is threatened by heavy metal sludges from a steel mill that are disposed of in unlined pits overlying the aquifer, as well as by a PCB (polychlorinated biphenyl) spill (Roberts *et al.*, 1982a; Schwartz *et al.*, 1982; Atwater, 1984). The outwash sand and gravel aquifers in the lower Fraser Valley of British Columbia are tapped by some 4000 operating wells and are threatened with contamination by nitrates, pesticides and landfill leachates. Similarly, the aquifer which forms the principal water supply for Prince Edward Island is threatened with contamination by aldicarb and other pesticides. Probably the worst case of ground-water contamination in Canada, however, occurs in the vicinity of a disposal site near Mercier, Quebec, where a sand and gravel aquifer tapped by 300 wells for domestic and agricultural purposes is contaminated by phenols and volatile chlorinated hydrocarbons (Simard, 1983).

For a substance to be an environmental hazard it must be toxic, persistent and mobile in the environment. It has been argued that the contaminants of greatest concern to American ground waters are synthetic organic chemicals because of their widespread occurrence in ground water, their toxicity and their mobility (Burmaster and Harris, 1982; Council on Environmental Quality [CEQ], 1981). Clearly, the same must apply to Canadian ground waters, especially in the heavily populated parts of southern Canada where surficial sand and gravel aquifers are an important component of the public water supply. It is these aquifers above all that must be protected from pollution by the large volumes of organic liquids and sludges containing toxic organic chemicals that are produced in Canada (Table 1).

Relatively little is known of the concentration of synthetic organic chemicals in Canadian ground waters. However, a recent analysis of ground waters taken from ten Canadian landfills indicated that 80% of the landfills surveyed were "not contaminating the local groundwater aquifers with significant concentrations of total organic halogen" (Zenon Environmental Inc., 1983). Since only three or four ground-water samples were taken at each landfill and a bailing technique was used for sampling, this conclusion may be optimistic as the present report shall show.

Table 1. Organic Wastes Generated during 1981 in Ontario

Waste category	Tonnes*
Organic sludges	43 000
Halogenated still bottoms	2 500
Degreasing sludge containing trichloroethane	28 500
Organic solvents and other organic solutions	42 100
Halogenated solvents	5 700
Non-halogenated solvents, e.g., benzene, acetone	22 500
Liquids containing PAH†	12 200

*1 tonne = 1000 kg = 220 imperial gallons or 1000 L aqueous solution.
†PAH – Polycyclic aromatic hydrocarbon.
Source: OWMC (1982).

THE GLOUCESTER LANDFILL

In 1957, Transport Canada leased approximately six hectares of land which is part of the Ottawa International Airport to the then Township of Gloucester for municipal waste disposal. The site, which will hereafter be referred to as the Gloucester Landfill, lies to the south of Ottawa and is situated on top of a complex sequence of fluvioglacial and littoral deposits of Quaternary age (Fig. 2). Between 1969 and 1980 various federal government agencies disposed of wastes in a "Special Waste Compound" at the

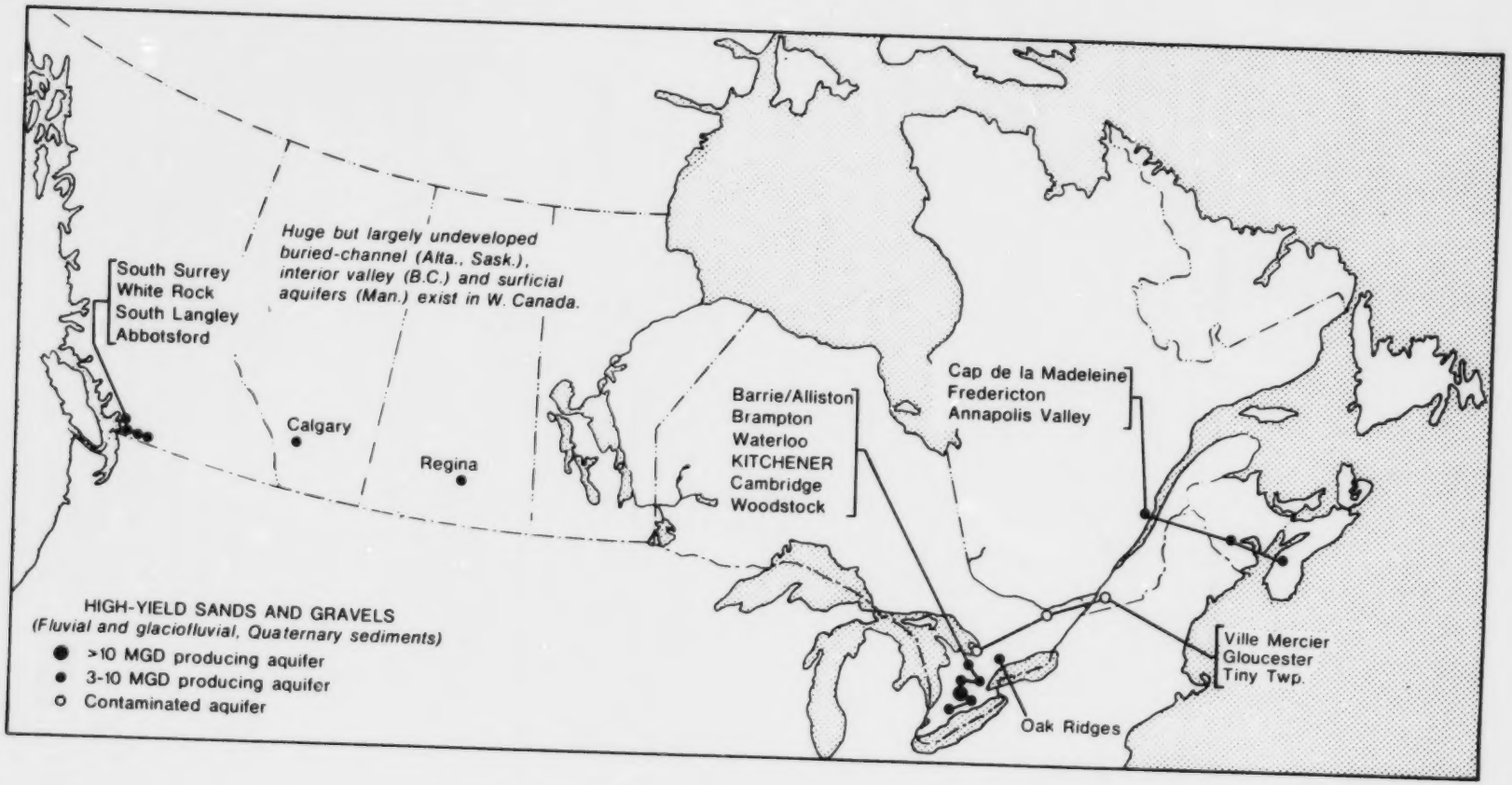


Figure 1. Sand and gravel aquifers of Canada.

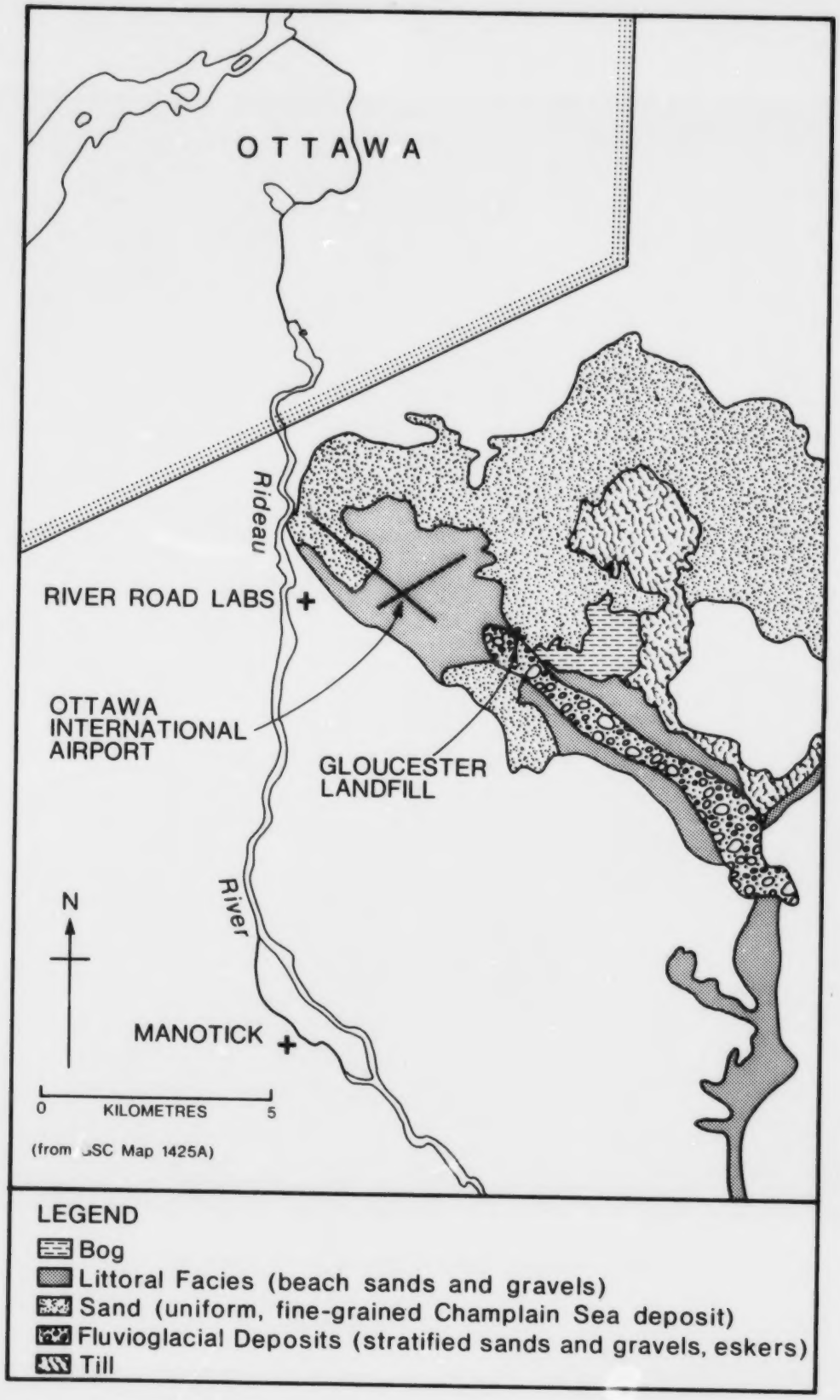


Figure 2. Location of the Gloucester Landfill and the Quaternary geology of the area.

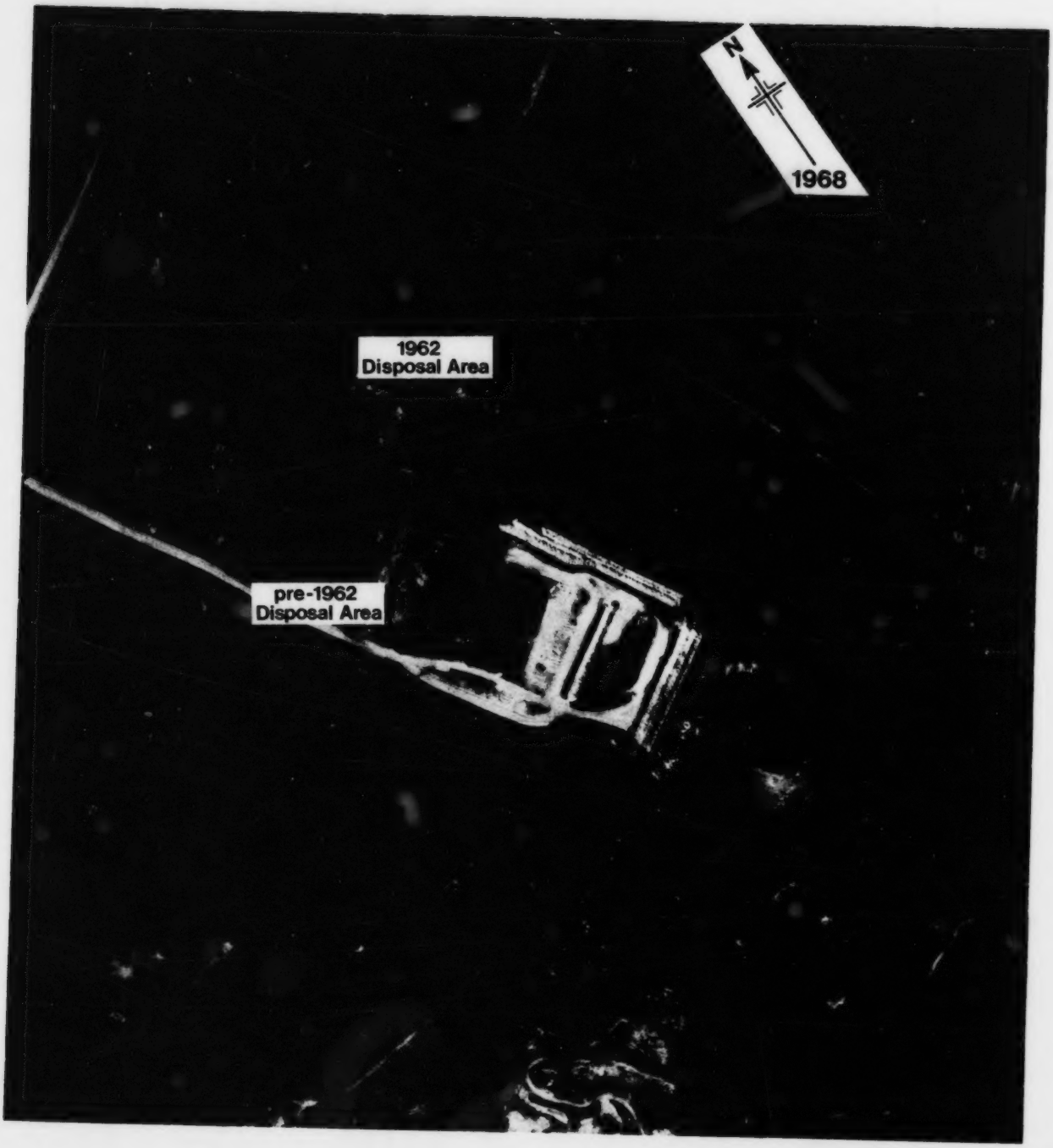


Figure 3. Aerial photograph of the Gloucester Landfill taken in 1968.

Figure 4. Aerial photograph of the Gloucester Landfill taken in 1973.

Table 2. Pollutants in Wastes Generated by Federal Government Facilities in the National Capital Region That May Have Been Disposed of at the Gloucester Landfill

Inorganic wastes	Organic wastes
Acids (nitric, hydrochloric, sulphuric, chromic)	Solvents (chloroform, carbon tetrachloride, methanol, acetone, benzene, hexane, etc.)
Bases (sodium hydroxide)	
Salts (mercuric, sodium citrate)	Pesticides (DDT, pentachlorophenol, parathion, Baygon, carbofuran, malathion)

Source: Gore and Storrie Ltd. (1977).

Landfill; these included hospital and university wastes. In the fall of 1979, scientists of the National Hydrology Research Institute (NHRI) of Environment Canada began a study of ground-water contamination at the site; this report summarizes the hydrogeological work to date, including an early survey of the site by Gartner, Lee Associates Ltd. (1978).

Aerial photographs are of considerable use in determining the history of waste disposal operations at the Gloucester Landfill. Figure 3 is an air photo taken in 1968 showing three open trenches dug with a dragline, containing waste materials and evidently penetrating the water table. The area to the north of the open trenches has been disturbed from previous disposal operations; the dates of use of this now abandoned area were obtained from other air photos which are not included in this report.

The second air photo (Fig. 4), taken in 1973, shows the final stages of the municipal disposal operations and a scar resulting from the federal disposals in the Special Waste Compound. A further point of interest in this photo is the depression to the east of the Special Waste Compound, where field inspection revealed a dug well. The darkened soil around this pit suggests that the area may have been used for the discharge of oily wastes which are known to have been disposed of (R. Jean, National Research Council, personal communication).

The early disposals (1969-1972) in the federal Special Waste Compound were undertaken by the National Research Council of Canada (NRCC), who would excavate a pit, 2.4 m x 2.4 m x 3.8 m, with a backhoe, place the wastes in the pit, and then ignite the wastes. However, it was reported in 1972 that "indiscriminate" dumping of hazardous wastes had taken place at the dump by certain unidentified federal departments. Subsequently, in December 1972, following a short period of operation by the Department of National Health and Welfare, Environment Canada took control of federal disposal operations at the former NRCC site. This resulted in redirection of much waste to Tricil in Mercier, Quebec, for incineration and also improved record-keeping of the wastes disposed of in the Special Waste Compound. Furthermore, Environment Canada followed an improved disposal procedure involving two trenches (12.2 m long x 3.0 m x 3.0 m) into which the wastes (mainly organic solvents in glass bottles) were placed and subsequently combusted following detonation of explosives set within the wastes. Very little residue was observable after such explosions and the trenches were then filled.

A wide variety of wastes are generated by federal government operations in the National Capital Region (i.e., Ottawa, Ontario, and Hull, Quebec). The wastes are summarized in Table 2. Of particular concern, because of their toxicity and mobility, are the large volumes of organic solvents originating in the various federal laboratories as well as hospital and university laboratories. For example, in one of the final disposals conducted by Environment Canada in May 1978, approximately one ton of organic solvents, in particular chloromethanes, was combusted (P. Mazzerole, Environment Canada, Hull, Quebec, personal communication). In addition, several special and unique disposals were made in the Compound that included the following pollutants: DDT, arsenic, cyanide, copper sulphate, hydrofluoric acid, metal carbonates and wood preservative solutions.

OBJECTIVES AND OUTLINE OF REPORT

The objectives of this study and their relevance to Environment Canada's goals are the following:

1. To develop and test sensitive field sampling and preservation and laboratory analysis procedures to characterize contaminants.

2. To assess the hydrogeological and geochemical controls on organic contaminant migration in the sand and gravel aquifer at the Gloucester Landfill and evaluate the applicability of predictive equations; this will result in an improved ability to evaluate the environmental impact of waste disposal practices and to conduct aquifer decontamination operations.

3. To develop and test remedial measures to contain, attenuate and/or remove contaminants from the aquifer; this will permit Environment Canada to intervene by proposing and/or conducting decontamination procedures at abandoned or operating waste disposal sites.

This report constitutes a summary of hydrogeological and chemical investigations conducted during the period 1979-1983 by NHRI to define the pattern of ground-water flow and contaminant migration in the ground-water flow system beneath the Gloucester Landfill with special emphasis on the Special Waste Compound. Therefore this document reports the Phase I activities as defined in NHRI's proposal of December 1980. Chapter 2 contains a discussion of toxic organic chemicals in terms of their physical chemistry, contaminant hydrogeology and health risks. Chapter 3 is a summary of the hydrostratigraphy of the site and contains a major contribution from J. Bahr of Geologic Testing Consultants (GTC) and Stanford University. Chapter 4 describes the physical nature of the ground-water flow system and is partly based on a study by D. Belanger, M.Sc. Project, University of Waterloo. The mineralogical, geochemical and hydrochemical aspects of the study are introduced in Chapter 5. The results of Phase I of the Gloucester Project are summarized in Chapter 6. Phase II will report on the mechanisms of organic pollutant migration, the numerical and experimental simulation of which are in preparation.

Toxic Organic Chemicals in Ground Water

Before any significant discussion of the contaminant hydrogeology of toxic organic chemicals can take place, it is necessary to consider briefly the chemical properties of organic compounds. In particular, the solubility, volatilization and partitioning of an organic chemical are of concern and are considered in the following section. Then contaminant migration and attenuation in ground water owing to solute transport, sorption and transformation are discussed. Because of these processes each compound will have its own characteristic migration velocity and concentration distribution. The final section of this chapter presents a method for quantifying the health risks of consuming chemically contaminated ground water.

PHYSICAL-CHEMICAL PROPERTIES OF ORGANIC CHEMICALS

Solubility

Many toxic organic chemicals are hydrophobic, that is, they are only sparingly soluble in water. The solubilities of selected organics are listed in Table 3; their structure is shown in Figure 5. If an organic compound is to dissolve in water, then the thermodynamic stability derived from the organic-water (i.e. solute-solvent) interactions must be greater than that from the molecular attraction of the organic solute molecules for one another (Spangler, 1980). For dissolution to occur, the organic solute must distort or disrupt the structure of water created by the strong hydrogen bonds between water molecules. If the organic solute contains hydrogen-bonding functional groups, such as hydroxy (-OH) and carboxy (-COOH) groups, and is therefore polar, then this is quite feasible and the organic solute will be relatively soluble in aqueous solution (Trahanovsky, 1971).

Tanford (1980) has described the process of solution of a nonpolar solute, S, in water schematically by

$$2(H_2O\text{-}H_2O) + (S\text{-}S) = 2(H_2O\text{-}S\text{-}H_2O)$$

The reaction driving this equilibrium is biased strongly to the left because of the energy of the hydrogen bonds between water molecules (5 kcal/mol in ice [Pauling, 1967]). Because of this, nonpolar solutes are sparingly soluble. The increased solubility owing to the addition of a polar functional group to a nonpolar solute can be appreciated by comparing the solubilities of methylbenzene (i.e. toluene) and phenol in Table 3. In Tanford's (1980) terminology, the "hydrophobic effect" which prevents the dissolution of large quantities of toluene is due to the energy required to disrupt the networks created by the hydrogen bonds and which form cagelike cavities within which nonpolar solutes such as toluene may be accommodated.

Volatilization

The factors that control the volatilization of organic compounds from ground water are (Lyman *et al.*, 1982; Tinsley, 1979)

1. The rate at which the compound migrates to a surface
2. The rate of transfer across the surface
3. The rate of migration away from this surface.

In ground water, the rate of migration of a toxic chemical to the top of the capillary fringe or to some part of the unsaturated zone from where it might evaporate is controlled by sorption, vertical ground-water flow and solute diffusion. The escaping tendency across an interface is a function of the vapour pressure of the compound. Migration of toxic chemicals from ground water to the ground surface is a function of the soil porosity and moisture content and the gaseous diffusion coefficient (Albertsen and Matthess, 1978).

Values of the vapour pressure and the Henry's law constant for selected organics are given in Table 3. The Henry's law constant of a compound may be approximated by dividing its vapour pressure by its solubility (Mackay and Leinonen, 1975). Therefore the constant may be thought of as a kind of partition coefficient with highly volatile substances having constants in excess of 10^{-3} atm-m^3/mol.

Halogenated Hydrocarbons

Dichloromethane (methylene chloride)	Trichloromethane (chloroform)	Tetrachloromethane (carbon tetrachloride)	Tribromomethane (bromoform)
CH_2Cl_2	$CHCl_3$	CCl_4	$CHBr_3$

1,1-Dichloroethane	1,2-Dichloroethane	1,1-Dichloroethene	Trans-1,2-Dichloroethene
Cl_2HC-CH_3	ClH_2C-CH_2Cl	$Cl_2C=CH_2$	$ClHC=CHCl$

1,1,1-Trichloroethane	Trichloroethene	1,1,2-Trichloroethane	Tetrachloroethene
H_3C-CCl_3	$Cl_2C=CHCl$	Cl_2HC-CH_2Cl	$Cl_2C=CCl_2$

1,1,2,2-Tetrachloroethane

$Cl_2HC-CHCl_2$

Aromatic Hydrocarbons

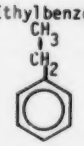
Benzene

Phenol

Methylbenzene

Chlorobenzene

Ethylbenzene



Other Organic Solvents

Acetone

Diethyl Ether

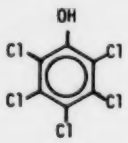
$H_3C-H_2C-O-CH_2-CH_3$

Tetrahydrofuran

1,4-Dioxane

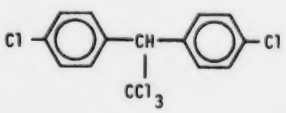
Biocides

Pentachlorophenol



DDT

(dichlorodiphenyltrichloroethane)



Lindane

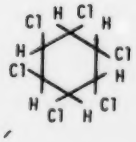


Figure 5. Molecular formulae of some of the organic compounds identified in ground waters beneath the Gloucester Landfill.

Table 3. Physical-Chemical Data of Selected Organic Chemicals

Compound	Formula	MW*	S†	V.P.‡	Henry's law§ constant	Log** K_{ow}
Halogenated hydrocarbons						
Dichloromethane	CH_2Cl_2	84.9	20 000 (20)	349	3×10^{-3}	1.25
Trichloromethane	$CHCl_3$	119.4	10 000 (15)	160	4.8×10^{-3}	1.97
Tetrachloromethane	CCl_4	153.8	800 (20)	90	2.3×10^{-2}	2.83
Bromoform	$CHBr_3$	252.8	3 190 (30)	5.6 (25°C)	5.8×10^{-4} c	2.38 f
1,1-Dichloroethane	$CHCl_2CH_3$	99.0	5 500 (20)	180	4.3×10^{-3} c	1.79
1,2-Dichloroethane	CH_2ClCH_2Cl	99.0	8 690 (20)	61	9.1×10^{-4} c	1.48
1,1-Dichloroethene	$H_2C=CCl_2$	97.0	–	500	–	2.13 f
Trans-1,2-dichloroethene	$CHCl=CHCl$	97.0	600 (20)	200 (14°C)	4.2×10^{-2} c	1.53 f
1,1,1-Trichloroethane	CCl_3CH_3	133.4	4 400 (20)	100	1.8×10^{-2}	2.49
Trichloroethene	$Cl_2C=CHCl$	131.5	1 100 (25)	60	1×10^{-2}	2.29
1,1,2-Trichloroethane	$CH_2ClCHCl_2$	133.4	4 500 (20)	19	7.4×10^{-4} c	1.51 f
Tetrachloroethene	$Cl_2C=CCl_2$	165.8	150 (25)	14	8.3×10^{-3}	2.60
Aromatic hydrocarbons						
Benzene	C_6H_6	78.1	1 780 (20)	76	5.5×10^{-3}	2.04
Phenol	C_6H_5OH	94.1	82 000 (15)	0.2	3.0×10^{-7} c	1.51
Methylbenzene	$C_6H_5CH_3$	92.1	470 (16)	22	6.6×10^{-3}	2.69
Chlorobenzene	C_6H_5Cl	112.6	500 (20)	8.8	2.6×10^{-3} c	2.84
Ethylbenzene	$C_6H_5C_2H_5$	106.2	140 (15)	7	8.7×10^{-3}	3.15
Other organic solvents						
Acetone	$CH_3-CO-CH_3$	58.1	M	89 (5°C)	–	-0.24
Diethyl ether	$C_2H_5OC_2H_5$	74.1	84 300 (15)	442	5.1×10^{-4} c	0.89
Tetrahydrofuran	C_4H_8O	72.1	M		–	0.46
1,4-Dioxane	$O(CH_2-CH_2)_2O$	88.2	M	30	–	-0.27
Biocides						
Pentachlorophenol	C_6Cl_5OH	266.4	14 (20)	1.1×10^{-4}	3.4×10^{-6}	5.12
DDT	$(ClC_6H_4)_2CHCCl_3$	354.5	0.0012 (25)+	1×10^{-7} +	3.8×10^{-5}	6.19
Lindane	$C_6H_6Cl_6$	290.8	7.3 (25)+	9.4×10^{-6} +	4.8×10^{-7}	3.61

* Molecular weight.

† Solubility in milligrams per litre at temperature in °C indicated in parentheses (Verschueren, 1977) or (+) (Kenaga and Goring, 1980); M – miscible in all proportions.

‡ Vapour pressure in millimetres Hg (torr) at 20°C (Verschueren, 1977).

§ Henry's law constant in atmospheres-cubic metres per mole (Lyman *et al.*, Table 15-4, 1982) or computed (c) from V.P. ÷ S (Mackay and Leinonen, 1975).

**Log of octanol-water partition coefficient from Hansch and Leo (1979) or (f) calculated from fragment constant method of Hansch and Leo (1979).

Octanol-Water Partitioning

During the past 50 years, many empirical correlations have been developed showing linear free-energy relationships between the structure and reactivity of organic compounds. In the past 20 years, this approach, known as "correlation analysis," has also been applied to environmental problems, including the partitioning of an organic contaminant in a ground-water flow system.

If the equilibrium constants for two chemical reactions, K_1 and K_2, are related by an expression of the form $y = ax + b$, then the relationship between the two reactions is known as a linear free-energy relationship because $\Delta G^\circ = -RT\ln K$ (Shorter, 1973). Since partition coefficients may be considered to be conditional equilibrium constants, it is possible to relate them by linear free-energy relationships to the physico-chemical properties of the chemicals being partitioned. Of particular interest is the relationship between the octanol-water partition coefficient, K_{ow}, and the water solubility of a chemical, S (Chiou *et al.*, 1982):

$$\log K_{ow} = -0.0862 \log S + 0.710$$

where $K_{ow} = C_o/C_w$, in which C_o is the equilibrium concentration of the chemical in octanol and C_w is its corresponding concentration in water in contact with the octanol solution. The term S is the molar solute solubility, which may be considered to be a measure of the partitioning of the chemical between itself and water (Briggs, 1981). Other correlations between K_{ow} and S and between K_{ow} and the bioconcentration factor or soil adsorption are given by

Kenaga and Goring (1980) and Briggs (1981). Values of K_{ow} are listed in Table 3 for selected organic chemicals. The reader is referred to Hansch and Leo (1979) for further details.

In terms of contaminant hydrogeology, the importance of the octanol-water partition coefficient is that it provides information on the likely behaviour of an organic compound in a flow system where organic matter is present as a sorbent.

SOLUTE TRANSPORT BY GROUND WATER

The average velocity of a nonreactive or conservative contaminant (i.e., one which does not undergo chemical or biological attenuation) is identical with that of the ground water transporting it (Bear, 1972, p. 23):

$$V_{GW} = q/n \qquad (2.1)$$

where V_{GW} is the average ground-water velocity (L/T), q is the specific discharge of ground water through a unit area of granular medium (L^3/L^2T), and n is the porosity (dimensionless).

Reactive contaminants are those whose average velocity is less than that of ground water because of their reaction (i.e. sorption) with aquifer materials. This reaction results in their distribution or partitioning between the aquifer grains and the ground water. Their average velocity (V_c) is given by (Freeze and Cherry, 1979, p. 404):

$$V_c = \frac{V_{GW}}{R_f} = \frac{V_{GW}}{1 + (\rho_b \ K_p/n)} \qquad (2.2)$$

where R_f is the dimensionless retardation factor, ρ_b is the bulk density of the granular medium (M/L^3), and K_p is the partition or distribution coefficient of the contaminant (L^3/M) as described by a linear adsorption isotherm (Tinsley, 1979, p. 18; Freeze and Cherry, 1979, p. 403).

The generalized equation which describes the mass balance of a contaminant with concentration $C(M/L^3)$ in uniform, steady flow in the x-direction in a granular medium is given by the transport equation (Wilson and Miller, 1978):

$$R_f \frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} + D_z \frac{\partial^2 C}{\partial z^2} - V_{GW} \frac{\partial C}{\partial x} - \lambda R_f C \qquad (2.3)$$

where t is time, D is the coefficient of hydrodynamic dispersion (L^2/T), and λ is a first-order rate constant (t^{-1}). As written above, the dispersion coefficient is a measure of the distribution of velocities about the mean value (i.e. V_{GW}) due to heterogeneities in the granular medium, drag forces exerted by the grains on the ground water, molecular diffusion, etc. The first-order rate constant may symbolize either a biotic or an abiotic (e.g. hydrolysis) transformation process.

SORPTION

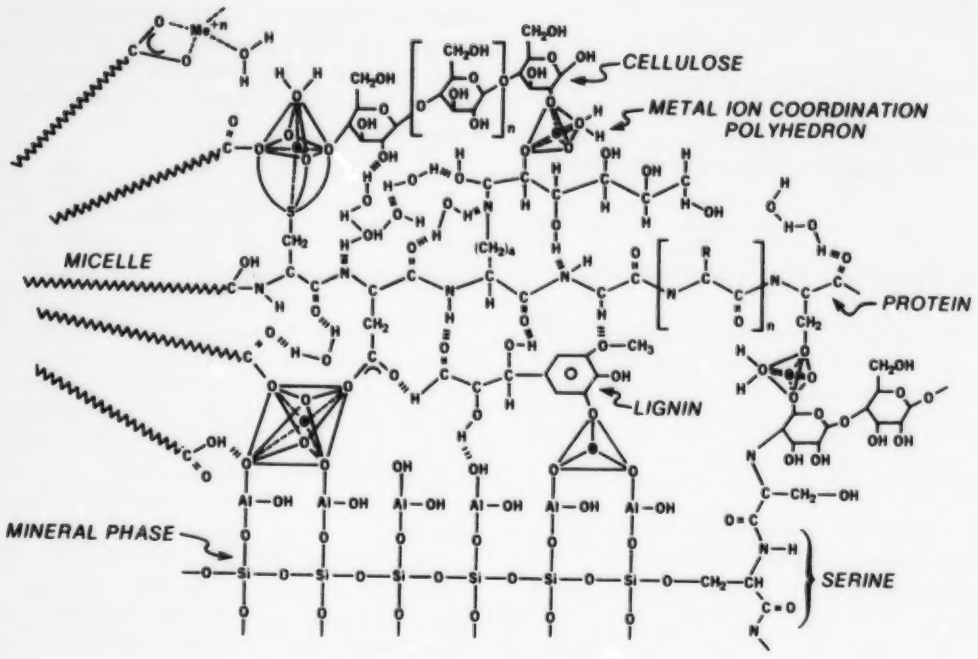
The Role of Organic Sorbents

The direct dependence of the uptake of neutral organic chemicals on the amount of soil organic matter became apparent in the mid-1960s, largely on account of the work of Lambert (1967). As a result of research by Chiou and colleagues at Oregon State University (1979, 1981, 1983) and Karickhoff *et al.* (1979) at the U.S. Environmental Protection Agency, a consensus has developed in America that uptake is the result of solute partitioning (i.e. due to dissolution in two or more phases) rather than physical or chemical adsorption (Roberts, 1981). However, the Israeli soil scientists Mingelgrin and Gerstl (1983) have argued that there exists "a continuum of possible interactions starting with fixed site adsorption and ending with true partition between three-dimensional phases." Consequently, since surface chemists (e.g. Aveyard and Haydon, 1973) use the term "sorption" to account for both adsorption and absorption (i.e. dissolution), this term will be used in the discussions which follow.

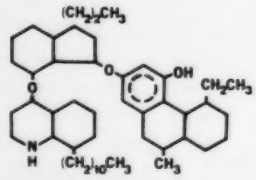
The essence of the partitioning hypothesis is based on the linearity of the sorption isotherms for compounds of widely differing solubilities and on the absence of competition between sorbates (see, for example, Chiou *et al.*, 1979, Fig. 1, and Chiou *et al.*, 1983, Fig. 1). Consequently, solute insolubility (i.e. in water) is perceived to be "the primary factor affecting the soil organic matter/water partition coefficient" (Chiou *et al.*, 1983). Chiou and colleagues (1979), however, concede that adsorption reactions (e.g. H bonding and ion exchange) may be significant for polar compounds. The sorption of organic solutes of low solubility is commonly referred to as hydrophobic bonding.

According to Chiou *et al.* (1983), "the effect of solute incompatibility with soil organic matter is significant but secondary." Chiou (1981) has noted that the poor retention of organic compounds by soil organic matter is likely the result of soil being composed of abundant highly polar polymers of large molar volume. The complex structure of these polymers is shown in Figure 6. In the

(A) SEDIMENTARY HUMIC SUBSTANCE

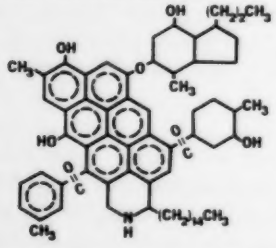


(B) ALGAL KEROGEN

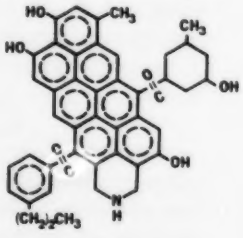


	B	C	D
ATOMIC H/C	1.65	1.28	0.84
ATOMIC O/C	0.06	0.10	0.13
HYDROCARBONS	OIL	OIL + GAS	GAS

(C) LIPTINITIC KEROGEN



(D) HUMIC KEROGEN



(E) SUBSTITUENT STRUCTURE OF KEROGEN

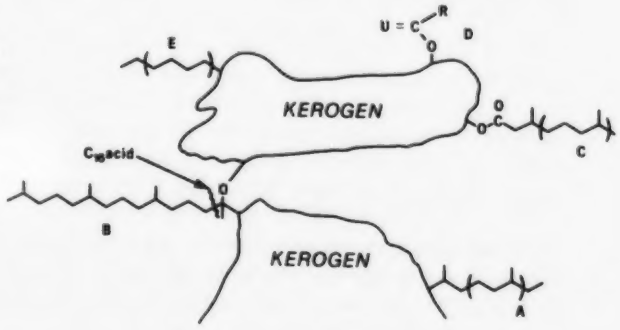


Figure 6. Schematic representations of organic matter: (A) sedimentary humic substance (Degens and Mopper, 1976); (B), (C) and (D) kerogen (Philip, 1982).

outwash sediments and soils of southern Ontario, it is likely that the organic fraction is largely composed of humic substances and kerogen, which is the insoluble portion of sedimentary organic matter. Most of this organic matter is concentrated in the fine-grained fraction (i.e. <125 μm) of soils and sediments (Karickhoff *et al.*, 1979; Schwarzenbach and Westall, 1981); consequently, organic-contaminant sorption is dependent on grain-size variations.

McCarty *et al.* (1981) reasoned that there must be a critical level of organic matter in aquifer materials at which sorption to organic and inorganic phases is identical, and below which the organic phase is no longer the dominant sorbent. They have calculated this level to be about 0.1% (f_{oc} = 0.001) for benzene ($K_{ow} \sim 10^2$) in contact with a silica surface of 13 m^2/g and about 0.001% (f_{oc} = 10^{-5}) for trichlorobenzene isomers with $K_{ow} \sim 10^4$. Because the organic carbon content of outwash aquifers is probably in the range of 0.01% to 1.0%, it is clear that inorganic adsorbents may more effectively retard some contaminants than do the organic sorbents. If inorganic sorbents dominate, then selective organic sorption is likely (McCarty *et al.*, 1981), particularly by amphoteric hydrous metal oxides.

Correlation Analysis

Many expressions have been developed to relate organic compound sorption to readily measured or available parameters. Karickhoff *et al.* (1979) showed that the sorption of an organic compound by soil organic matter could be expressed by:

$$\log K_{oc} = 1.00 \log K_{ow} - 0.21 \quad (r^2 = 1.00) \quad (2.4)$$

where K_{oc} is the partition coefficient between soil organic carbon and water, i.e. (g solute sorbed/g soil organic carbon)/(g solute/m^3 solution). It is related to the partition or distribution coefficient by the relation $K_p = K_{oc}\, f_{oc}$, where f_{oc} is the weight fraction of organic carbon in the soil material.

Schwarzenbach and Westall (1981) used a linear free-energy relationship to relate the partition coefficients of a nonpolar solute z in different aqueous/nonaqueous solvent systems:

$$\log K^z_{SW} = a \log K^z_{RW} + b \quad (2.5)$$

where K^z_{SW} is the partition coefficient of z between the solvent S and water and K^z_{RW} is that for a reference solvent R such as octanol. The slope constant is a measure of the lipophilicity of solvent S relative to R, and hence, when the solvent is sedimentary organic matter, it is a measure of its capacity for sorption. For compounds $2.6 < \log K_{ow} < 4.7$ in contact with alluvial aquifer materials, Schwarzenbach and Westall found that the value of a varied from 0.50 to 0.71. The intercept constant is a function of the units in which K is measured.

Schwarzenbach and Westall went on to describe the sorption of chlorinated benzenes by sediments with $f_{oc} > 0.001$ by the following relationship:

$$\log K^z_p = 0.72 \log K^z_{ow} + \log f_{oc} + 0.49 \quad (r^2 = 0.95) \quad (2.6)$$

where K^z_p is the partition (or distribution) coefficient of z in units of millilitres per gram. This value of the partition coefficient may be used in the retardation equation (Equation 2.2) to estimate the retardation factor of an organic contaminant in a porous medium of known porosity and bulk density; the relationships between K_{ow} and R_f and between K_{ow} and the relative ionic velocity (V_c/V_{GW}) are shown in Figure 7.

Comparisons are shown in Table 4 of observed R_f values from two Stanford University field tests and the computed values estimated from Figure 7 and denoted as R^*_f. With the exception of chlorobenzene (which was used in developing Equation 2.6), the estimated values are

Table 4. Retardation Factors for Selected Halogenated Hydrocarbons from Field Observations (R_f) and Estimated (R^*_f) from Equations 2.2 and 2.6

Compound	R^*_f	R_f	Statistics* of field data	Aquifer
$CHCl_3$	8.4	2.5–3.8	$\bar{x}$ = 3.2, σ = 0.6, s = 4	Palo Alto Baylands†
$CHBr_3$	16	6	–	Palo Alto Baylands
Cl_3CCH_3	18	12	–	Palo Alto Baylands
C_6H_5Cl	32	33	–	Palo Alto Baylands
$CHBr_3$	1.1	1.5–1.8	$\bar{x}$ = 1.7, σ = 0.2, s = 3	Borden‡
CCl_4	1.3	1.4–1.7	$\bar{x}$ = 1.6, σ = 0.2, s = 3	Borden

* $\bar{x}$ = mean, σ = standard deviation, s = sample size.
† R_f data from Roberts *et al.* (1982b); ρ_b = 2000 kg/m^3, n = 0.22, f_{oc} = 0.01 (McCarty *et al.*, 1981).
‡ R_f data, ρ_b = 1760 kg/m^3, n = 0.38, f_{oc} = 0.000 75–0.002 (Mackay *et al.*, 1983); f_{oc} assumed = 0.0002 in R^*_f calculation.

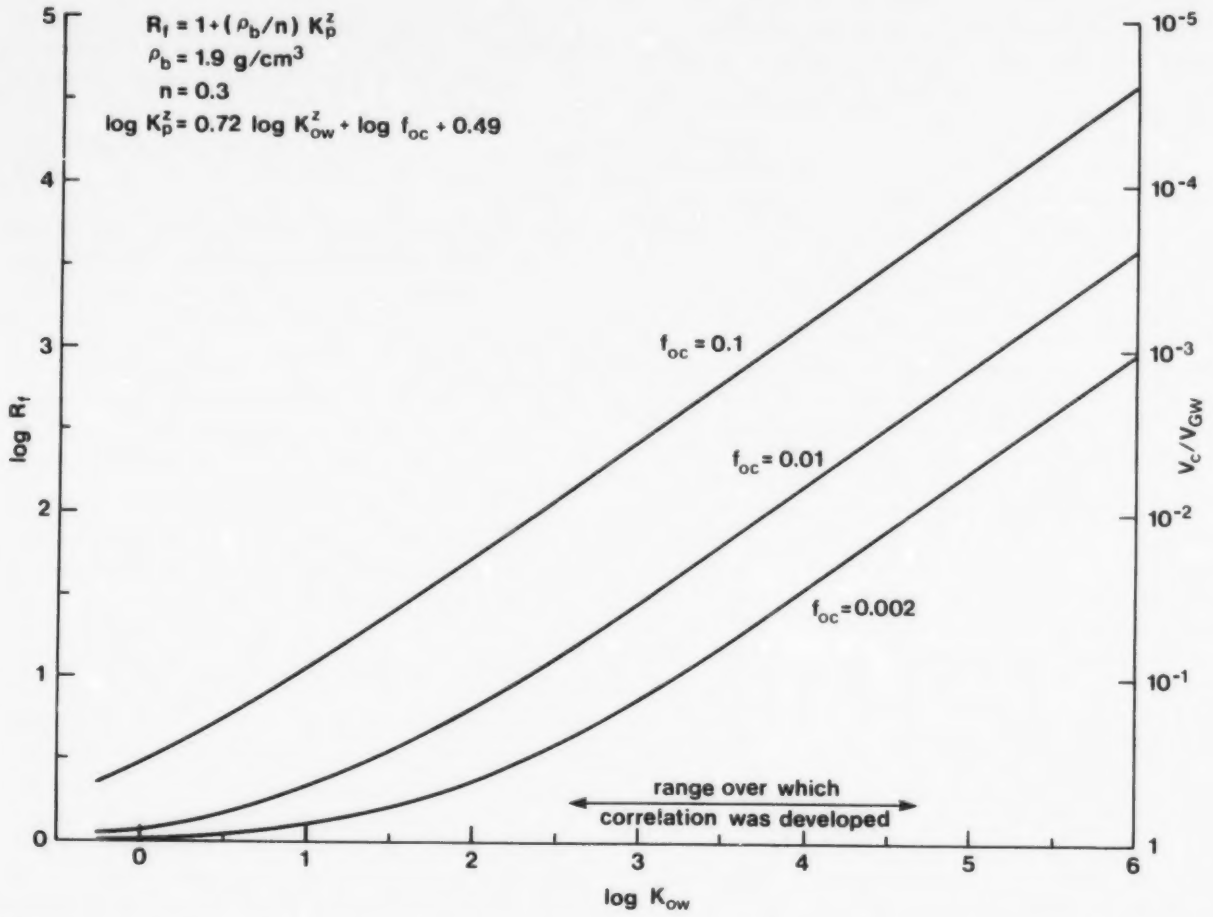


Figure 7. Variation of the retardation factor with the octanol-water partition coefficient for various concentrations of sedimentary organic carbon using the equation of Schwarzenbach and Westall (1981).

greater by factors of 1 to 3 than the observed values. Given that the concentrations of the organic compounds in these field studies were in the low ppb levels (1-40 μg/L) and therefore similar to the experimental conditions of Schwarzenbach and Westall (1981), the differences are somewhat disturbing. If Equation 2.4 (Karickhoff *et al.*, 1979) had been used instead of that of Schwarzenbach and Westall (1981), as has been done by Roberts and Valocchi (1981), then improved estimates of R_f would have been obtained for the more soluble compounds, e.g., chloroform ($R_f^* = 6.3$, log $K_{OW} = 1.97$), but a poorer estimate would result for a less soluble compound such as chlorobenzene ($R_f^* = 40$, log $K_{OW} = 2.84$).

Recently, Curtis and Roberts (1984) have demonstrated good agreement between laboratory-estimated retardation factors for CCl_4, $CHBr_3$ and C_2Cl_4 and estimates from field observations taken during a natural-gradient, organic solute transport experiment at CFB Borden, Ontario. They concluded that linear, reversible sorption was therefore occurring in the Borden aquifer. They noted that a period of two to three days was required for the compound to reach equilibrium with the Borden aquifer material during batch K_p experiments.

The methods mentioned above permit approximate estimates of contaminant mobility under conditions of low-level contamination, such as at the front of an advancing plume. In the immediate vicinity of a landfill, however, contaminant concentrations will likely be substantially higher and estimates of contaminant retardation significantly more uncertain. In heavily contaminated porous media, sorption isotherms may be strongly nonlinear and microbial activity much reduced, resulting in greater mobility of the contaminant (Rao and Davidson, 1979). As the fraction of miscible organic solvent to ground water increases in the contaminated pores, the retardation factor exponentially approaches unity (Rao *et al.*, 1983).

TRANSFORMATION

Most of the toxic organic contaminants found in ground water owe their hazardous properties to halogen atoms, in particular to chlorine. In this section recently published evidence for the transformation of halogenated compounds and a few hazardous but nonhalogenated compounds (e.g. benzene) is summarized.

It should be noted that some transformation processes depend only upon the physical and chemical properties of the ground water (e.g. pH, E_H, T) to proceed; these relatively slow processes are usually hydrolysis reactions in that they involve a component ion of the water molecule. Many other transformation processes, however, occur more rapidly owing to their catalysis by microbially produced enzymes; these reactions are commonly referred to as biodegradation processes or microbial or metabolic transformations. In this latter case, it is probable that the principal constraints on microbial proliferation and enzymatic function are ground-water temperature, redox conditions and nutrient availability (Champ *et al.*, 1979).

Organic Reaction Mechanisms

Irrespective of whether or not a transformation process is microbially mediated, all organic reaction mechanisms may be described by one or more of four general reaction types (Sykes, 1981): substitution, elimination, addition and rearrangement. The transformations of halogenated organic compounds are mainly confined to substitution and elimination reactions.

The classic transformation of an alkyl halide into an alcohol is due to the substitution of the halide (Hal) by the hydroxyl ion (OH^-):

$$OH^- + R\text{-}Hal \rightarrow HO\text{-}R + Hal^- \qquad (2.7)$$

where R refers to the alkyl group. Because the hydroxyl ion is a nucleophile or electron-pair donor and if, as is frequently the case, the reaction rate is a function of the concentration of both reagents (i.e. OH^-, R-Hal), then this mechanism is known as "substitution nucleophilic bimolecular," or S_N2. If the rate of this reaction is dependent only upon the concentration of one reagent, i.e. R-Hal, it is classified as S_N1.

Elimination reactions provide a second mechanism by which organic halides may undergo dehalogenation, thereby releasing to solution the inorganic halide ion (Hal^-). The classic reaction of this kind is the base-induced elimination of hydrogen halide from alkyl halides, in particular from alkyl bromides, to form an alkene and free halide ion (Sykes, 1981, Chapter 9):

$$RCH_2CH_2Br \xrightarrow{-OH} RCH = CH_2 + H_2O + Br^- \qquad (2.8)$$

Because the rate of the reaction is controlled by the concentration of both the base and the substrate, i.e.

$$\text{Rate} = k[RCH_2CH_2Br][-OH] \qquad (2.9)$$

where k is the rate constant, it is classified as an E2 mechanism (elimination, bimolecular). Since the base is a nucleophile, the E2 reaction is often accompanied by an S_N2 reaction.

Although physical organic chemistry provides a framework within which environmental transformation processes may be interpreted, environmental chemists generally describe halogen transformations in terms of three principal reaction routes (e.g. Hill, 1978; Tinsley, 1979): hydrolysis, dehydrohalogenation and reductive dehalogenation. These routes, which are shown in Figure 8, may be either abiotic or microbially mediated.

Hydrolysis (Abiotic Transformation)

Mabey *et al.* (1983) have pointed out that several polyhalogenated alkanes of environmental interest undergo OH^--induced hydrolysis to yield an alkene and HHal by dehydrohalogenation. Examples of compounds which undergo this E2 process include the trihalomethanes (Mabey and Mill, 1978); the pesticide 1,2-dibromo-3-chloropropane (Burlinson *et al.*, 1982); and a number of chlorinated ethanes (Walraevens *et al.*, 1974). However, the rates of these reactions at pH 7 are slow. The following method of quantification of these rates is taken from Mabey and Mill (1978).

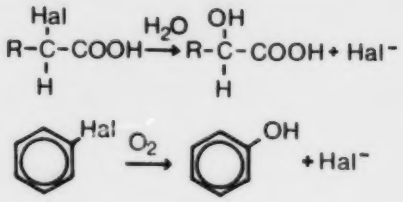
For the hydrolysis of compound RX involving the displacement of X by OH^- at the reaction centre, i.e.

$$RX + H_2O \rightarrow ROH + HX \qquad (2.10)$$

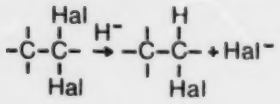
the rate law is written

$$d(RX)/dt = k_h[RX] = k_b[OH^-][RX] + k_a[H^+][RX] + k_n[H_2O][RX] \qquad (2.11)$$

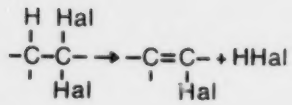
where k_b, k_a and k_n are second-order rate constants for the base-catalyzed, acid-catalyzed and neutral processes, respectively, and k_h is a pseudo first-order rate constant at a constant pH. Assuming that the individual rate processes



HYDROLYTIC DEHALOGENATION OF ALIPHATIC (S_N2 WITH OH^-) AND AROMATIC (S_NAr) COMPOUNDS



REDUCTIVE DEHALOGENATION (S_N 2)



DEHYDROHALOGENATION (E2 WITH OH^-)

Figure 8. Reactions important in the transformation of organic pollutants (after Hill, 1978).

for the acid, base and neutral hydrolyses are first order with respect to the substrate, RX, then

$$k_h = k_b[OH^-] + k_a[H^+] + k_n \qquad (2.12)$$

Furthermore, because there are no known acid-promoted hydrolysis reactions for halogenated alkanes (Mabey *et al.*, 1983)

$$k_h = k_b[OH^-] + k_n \quad (T \text{ constant}) \qquad (2.13)$$

At a fixed pH (or pOH), the overall rate process is pseudo first-order and the half-life of the substrate, RX, is therefore independent of its concentration:

$$t_{1/2} = 0.693/k_h \qquad (2.14)$$

The hydrolysis rates and half-lives of several halogenated compounds are given in Table 5. It should be noted that these rates and half-lives are for 25°C and pH 7; Mabey and Mill (1978) estimate that at 14°C, the half-lives are 130% longer.

Not all hydrolysis reactions involve E2 dehydrohalogenation. Walraevens *et al.* (1974) reported that monochloroethane, 1,1-dichloroethane and 1,1,1-trichloroethane undergo slow substitution reactions (probably S_N2). However, 1,2-dichloroethane hydrolyzes by both E2 elimination and S_N2 substitution.

It must be pointed out that variations in experimental conditions can result in different reaction products (i.e. E vs. S_N; March, 1977, pp. 914-917). For example, the S_N2 substitution of 1,1,1-trichloroethane (IIITCEA) reported by Walraevens *et al.* (1974) must produce 1,1-dichloro-1-ethanol. In contrast, McConnell *et al.* (1975) reported that the dehydrochlorination of IIITCEA in sea water at 10°C and pH 8 yields 1,1-dichloroethane with $t_{1/2}$ = 9 months. Alternatively, Mabey *et al.* (1983) indicate that at 25°C "only acetic acid (CH_3COOH) and chloride ion were found as hydrolysis products" of IIITCEA. However, they agreed on the value of $t_{1/2}$ and noted that the rate constants were independent of pH.

Finally, it is worth noting that nucleophiles other than OH^- are capable of reacting with alkyl halides via substitution reactions. Schwarzenbach *et al.* (1983) have shown that at pH 8 and $[H_2S]_{TOT} \sim 5 \times 10^{-5}$ M (~2 ppm total H_2S) volatile organic sulphur compounds, such as alkylmercaptans, can form by the substitution of HS^- for Br^-. Sulphur compounds of this kind were identified beneath a leaking wastewater tank at a chemical plant in Switzerland and may well occur whenever anaerobic conditions cause significant sulphate reduction in the presence of alkyl halides.

Table 5. Rates of Hydrolysis and Half-lives of Halogenated Compounds

Compound	Rate constants (1/s) at pH 7 and 25°C			$t_{1/2}$	References
	k_n	$k_b[OH^-]$	k_h		
CH_2Cl_2	3.2×10^{-11}	2.1×10^{-13}	3.2×10^{-11}	704 yr	Mabey and Mill (1978)
$CHCl_3$	–	6.9×10^{-12} •	6.9×10^{-12} •	3500 yr	Mabey and Mill (1978)
$CHBr_3$	–	3.2×10^{-11} •	3.2×10^{-11} •	686 yr	Mabey and Mill (1978)
CCl_4	–	4.8×10^{-7} †	4.8×10^{-7} †	7000 yr	Mabey and Mill (1978)
$1,1,1\text{-}C_2H_3Cl_3$	3.4×10^{-8}	–	–	8 months	Mabey *et al.* (1983)
$1,2\text{-}C_2H_4Cl_2$	4×10^{-10}	–	–	55 yr (pH 5)	Mabey *et al.* (1983)

•It is assumed that $k_h = k_b[OH^-]$.
†Hydrolysis is second order in CCl_4 (k_h:litre/mole × second); $t_{1/2}$ is instantaneous value at 1 ppm.

Table 6. Evidence for the Transformation of Volatile Organic Contaminants under Different Redox Regimes

Compound	Aerobic ($E_H \geq 0.4$ V)	Denitrification ($0.4 \text{ V} \leq E_H \leq 0.0$ V)	Methanogenic ($E_H \leq 0.0$ V)
Halomethanes			
$CHCl_3$	I F	I F′	P F′
$CHBr_3$	I	P F′	P F′
CCl_4	I	P F′	P F′
BDCM	I	P F′	P F′
DBCM	I	P F′	P F′
Halogenated ethanes and ethenes			
1,2-DCEA	I	N.D.	P
1,2-DBEA	I	I	P
Tri CEE	N.D. F	N.D.	P
1,1,1-CEA	I F	I	P
Tetra CEE	I F	N.D.	P
1,1,2,2-CEA	I	N.D.	P
Chlorobenzenes			
Chlorobenzene	P	I	I
1,2-DCB	P	I	I
1,3-DCB	P	I	I
1,4-DCB	P F	I	I
1,2,4-TCB	P	I	I

P – Transformation probable.
I – Transformation improbable.
F – Probability or improbability supported by field evidence.
F′ – Field evidence supports this, but although the redox environment was anaerobic, the E_H was not known.
N.D. – No data.

Biodegradation

Recent studies of the microbial transformation of the common organic pollutants of ground water suggest that the redox (oxidation-reduction) environment is all important. Assuming that ground-water temperature and dissolved nutrient concentrations are capable of supporting microbial growth and enzymatic function (Champ *et al.*, 1979), then it is likely that aromatic compounds will be biodegraded under oxic (i.e. aerobic) conditions, whereas halogenated aliphatic compounds will be biotransformed only in anoxic environments (McCarty *et al.*, 1984). The potential for biodegradation of various organic pollutants in different redox environments is summarized in Table 6.

Oxygenated ground waters are likely to be found in the recharge areas of granular aquifers (Champ *et al.*, 1979; Jackson and Patterson, 1982) but, on occasion, may be observed at depth within ground-water flow systems (Winograd and Robertson, 1982). In such oxic environments aromatic hydrocarbons, such as benzene, toluene, phenols, chlorobenzenes and chlorophenols, are likely to

undergo biotransformation (Wood, 1982; Wilson and McNabb, 1983; Bouwer, 1983). In anoxic conditions if oxygen is not present on the aromatic ring or in a substituent of the ring (e.g. OH^-, COOH), then the cleavage of the ring (i.e. biodegradation) appears unlikely (Bouwer and McCarty, 1983b). In recent laboratory studies involving biofilm columns, a variety of chlorobenzenes (~10 ppb each) were cometabolized with acetate (1 ppm) under aerobic conditions but not under methanogenic conditions (Bouwer, 1983; Bouwer and McCarty, 1983b). Marinucci and Bartha (1979) also reported that trichlorobenzenes were not anaerobically biodegraded.

Field studies of aromatic hydrocarbons by the U.S. Geological Survey and EAWAG (the Federal Institute for Water Resources and Water Pollution Control, Dubendorf, Switzerland) have produced data which are consistent with the foregoing statements. Ehrlich *et al.* (1982) have shown that phenolic compounds undergo anaerobic biodegradation in ground waters containing 2-20 mg CH_4/L; because phenols contain ^-OH groups this observation appears to confirm the conclusion of Bouwer and McCarty (1983b) concerning the importance of oxygen-containing functional groups in the anoxic biotransformation of aromatic compounds. Schwarzenbach *et al.* (1983) observed the biotransformation of alkylated and halogenated aromatic compounds under oxic conditions but, significantly, noted that 1,4-dichlorobenzene appeared not to be biotransformed during periods of denitrification within the aquifer.

Once the available molecular oxygen in ground water has been reduced, redox reactions result in the sequential reduction of nitrate, manganese and iron oxides, sulphate and dissolved carbon dioxide (Champ *et al.*, 1979; Jackson and Patterson, 1982). Therefore denitrification conditions are mildly reducing compared with environments within which sulphate reduction and methane fermentation occur. In reducing environments, reductive dehalogenation and dehydrohalogenation reactions are known to biodegrade halogenated aliphatic compounds. The best documented occurrence of these reactions is the biotransformation of DDT to TDE by reductive dechlorination and of DDT to DDE by dehydrochlorination (Hill, 1978). Esaac and Matsumura (1980) note that the rate of DDT biotransformation appears to be a direct function of the "soil redox potential."

From laboratory studies it appears that halogenated aliphatics undergo anaerobic dehalogenation by processes similar to DDT degradation. Bouwer and McCarty (1983a,b) showed that with an acclimation period, carbon tetrachloride and three brominated trihalomethanes were degraded in the presence of denitrifying bacteria. Furthermore, chloroform, carbon tetrachloride and 1,2-dichloroethane were almost completely transformed to CO_2 by biological oxidation under methanogenic conditions. They also presented evidence for the reductive dechlorination of tetrachloroethylene and 1,1,2,2-tetrachloroethane under methanogenic conditions. Parsons *et al.* (1984) have also reported the reductive dechlorination of chlorinated aliphatics in laboratory anaerobic environments. Likewise several halogenated aliphatic pesticides, such as ethylene dibromide and 1,2-dibromochloropropane, are reductively dehalogenated (Castro and Belser, 1968), although a dehydrochlorination reaction may be responsible for the removal of the second halogen atom (Hill, 1978).

The results of field studies in Europe and California are consistent with the foregoing conclusions. Biotransformation of trihalomethanes ($t_{1/2}$: 3-6 weeks) and chlorinated ethanes and ethenes ($t_{1/2}$: 5-9 months) was observed in an anoxic aquifer receiving reclaimed municipal wastewater (Bouwer, 1983). Piet and Zoeteman (1980) also reported that trihalomethanes (20-100 ppb) are "eliminated" under anoxic conditions during passage through sand dunes. However, Schwarzenbach *et al.* (1983) concluded that chloroform, 1,1,1-trichloroethane, trichloroethene and tetrachloroethene were persistent under oxic and seasonally anoxic conditions during infiltration of river water to an alluvial aquifer.

In summary, therefore, there is a growing body of research which indicates that in oxic conditions aromatic compounds but not aliphatic halides will be biotransformed. In anoxic conditions halogenated aliphatics and only aromatics with functional groups containing oxygen may undergo biodegradation. However, the constraints on the microbial transformation of these compounds owing to ground-water temperatures, nutrient availability, acclimation period and redox conditions are not well understood, and until they are, the prediction of biotransformation processes *in situ* will be subject to much uncertainty.

HEALTH RISKS

Many of the organic compounds disposed of at the Gloucester Landfill (Table 2) are known to cause or are suspected of causing health problems. For example, benzene is a confirmed human carcinogen and has been associated with prolonged menstrual bleeding, and several chlorinated ethanes and ethenes (ethylenes) are confirmed animal carcinogens (CEQ, 1981). Recommended drinking water guideline values have been proposed by the World Health Organization (WHO) for only a small number of the organic chemicals that have been identified as posing health risks (Table 7).

Table 7. Evidence for Carcinogenicity, Drinking Water Guideline Values and Cancer Risks for Selected Toxic Organic Chemicals Disposed of at the Gloucester Landfill

Chemical	Evidence for carcinogenicity*	Drinking water guideline† (μg/L)	Cancer risk‡
Benzene	H	10	4.4×10^{-6}
Carbon tetrachloride	A	3	1.9×10^{-6}
Chloroform	A	30	4.1×10^{-6}
1,1-Dichloroethane	S.A.	—	1.5×10^{-4}
1,2-Dichloroethane	A	10	1.0×10^{-6}
1,1-Dichloroethylene	N.T.	0.3	—
Lindane	A	3	1.3×10^{-5}
Tetrachloroethylene	A	10	9.3×10^{-7}
Trichloroethylene	A	30	3.0×10^{-7}

*H – confirmed human carcinogen, A – confirmed animal carcinogen, S.A. – suggested animal carcinogen, N.T. – not tested in animal bioassay. Source: CEQ (1981).
†Recommended and, in some cases, tentative WHO drinking water guideline values, 1984. N.B. 1978 Canadian Drinking Water Quality guidelines recommended 4 μg lindane/L and 350 μg trihalomethane/L.
‡Estimated upper statistical confidence limit (95%) on cancer risk from lifetime consumption of water containing 1 ppb of the specific chemical. Source: CEQ (1981).

Although concentrations recommended as drinking water guidelines indicate the potential hazard of a given substance—100 ppb for benzene, 200 ppb for cyanide, 1000 ppb for copper—they do not permit an assessment of the risk involved in drinking that substance. However, a method has been developed on behalf of the CEQ (Council of Environmental Quality, Washington, D.C., U.S.A.) that is based on evidence from (1) occupational exposure studies, (2) epidemiological studies of populations drinking chlorinated water, and (3) laboratory studies of chemical carcinogens. This method permits the calculation of a risk assessment, albeit a crude one (CEQ, 1981).

The right-hand column of Table 7 presents the upper confidence limits for human cancer risk of drinking water containing 1 ppb of a specific organic chemical. The human being concerned is assumed to weigh 70 kg and consume 2 L of water per day; animal risks were converted to human risks by a surface area conversion procedure. Therefore the risk of consuming 1 ppb benzene over a lifetime is 4.4×10^{-6}, i.e., should one million people consume water containing 1 ppb benzene over a lifetime, 4.4 would contract cancer over the course of their life from this cause alone. Assuming a linear dose-incidence relationship, it is possible to multiply the risk due to a lifetime dose of 1 ppb by the average concentration of the chemical in ground water to arrive at an estimate of the upper confidence limit for the average concentration. Furthermore, assuming that risks from specific chemicals are additive, the net risk of incurring cancer from drinking contaminated ground water may be calculated by adding the separate risks from each specific chemical.

Using this "multistage" method to estimate potential lifetime cancer risks for contaminated ground water in New Jersey (230 ppb benzene, 400 ppb CCl_4, 1500 ppb trichloroethene), the CEQ (1981, p. 70) estimated a risk of 2.59×10^{-3}, or approximately 1 in 400. This value is substantially higher than is generally considered acceptable by U.S. state and federal agencies, i.e. 10^{-6}, or 1 in 1 million. An estimate of the risk involved in drinking contaminated ground water from the Gloucester Landfill is given in Chapter 5.

Hydrostratigraphy

REVIEW

The hydrostratigraphic interpretation for the Gloucester site evolved throughout the course of the investigation as new information became available. In many instances, the later data not only added detail but indicated that reinterpretation of some of the earlier findings was necessary.

The first hydrostratigraphic investigation at the site was carried out by Gartner, Lee Associates (1978). A second study was completed by French and Rust (1981) and involved the drilling of eight boreholes with split spoon samples being collected at 1.5- or 3-m intervals. The results indicated highly variable stratigraphic and sedimentologic relationships, and it was recognized that interpretations could only be made with caution because (1) critical contacts may have been missed, (2) fossils are absent, and (3) the sampling method provided little or no information about sedimentary structures. With these qualifications, most of the sediments were interpreted as subaqueous esker-fan deposits. Only one unit in the whole sequence was considered to be definitely of marine origin, and many strata were believed to be mass flow deposits.

J. Bahr, Geologic Testing Consultants Ltd. (GTC), completed a detailed, three-dimensional hydrostratigraphic interpretation of the unconsolidated deposits at the site based on data available in 1982. This interpretation included not only information on site geology but also data on water chemistry and hydraulic properties. The most recent reinterpretation by NHRI incorporates new stratigraphic data obtained during a drilling program carried out in 1983.

Throughout the course of site investigation, studies concerned with contaminant migration and hydraulic modelling were conducted, and in each case, the most recent hydrostratigraphic interpretation was used. Since the GTC hydrostratigraphic model was employed extensively, the details of this interpretation, in addition to the most recent NHRI reinterpretation, are included in the following sections.

THE HYDROSTRATIGRAPHIC MODEL OF GTC

Regional Setting

The Gloucester Landfill lies on the eastern flank of a northwest-southeast trending sand and gravel ridge, south of the Ottawa International Airport. Several published maps identify this ridge as glaciofluvial deposits (Gadd, 1963; Richard, 1976). More recently, this ridge has been interpreted as consisting of subaqueous outwash deposited close to the ice front but below wave base in the Champlain Sea and/or an earlier ice dammed lake (Rust and Romanelli, 1975; Rust, 1977). According to the most recent surficial map of the Ottawa region published by the Geological Survey of Canada (Belanger and Harrison, 1980), surficial deposits to the east of the ridge comprise littoral and sublittoral facies of the Champlain Sea. The ridge and adjacent sediments were probably reworked to some extent during the retreat stages of the Champlain Sea.

Basis of Interpretation

A hydrostratigraphic interpretation of the area surrounding the special waste site is illustrated by a fence diagram as shown in Figure 9. This diagram is drawn with no vertical exaggeration. The base of the fence posts is at a datum of 68 m above mean sea level.

Identification and correlation of sediments and their stratigraphic relationships required a synthesis of data from a variety of sources. Detailed logs from exploratory borings were available for only a limited number of locations. Sediment samples recovered during piezometer installation provided additional information and served as a check on the drilling records. Several test pits excavated above the water table permitted examination of the sedimentary structures of the uppermost unit. Finally, the relative hydraulic conductivities of sediments at various depths were evaluated during a water quality sampling program conducted during the summer and autumn of 1982. At several locations for which reliable stratigraphic information was available from drill logs and samples, it was noted that the relative production rates of the sampling points within a

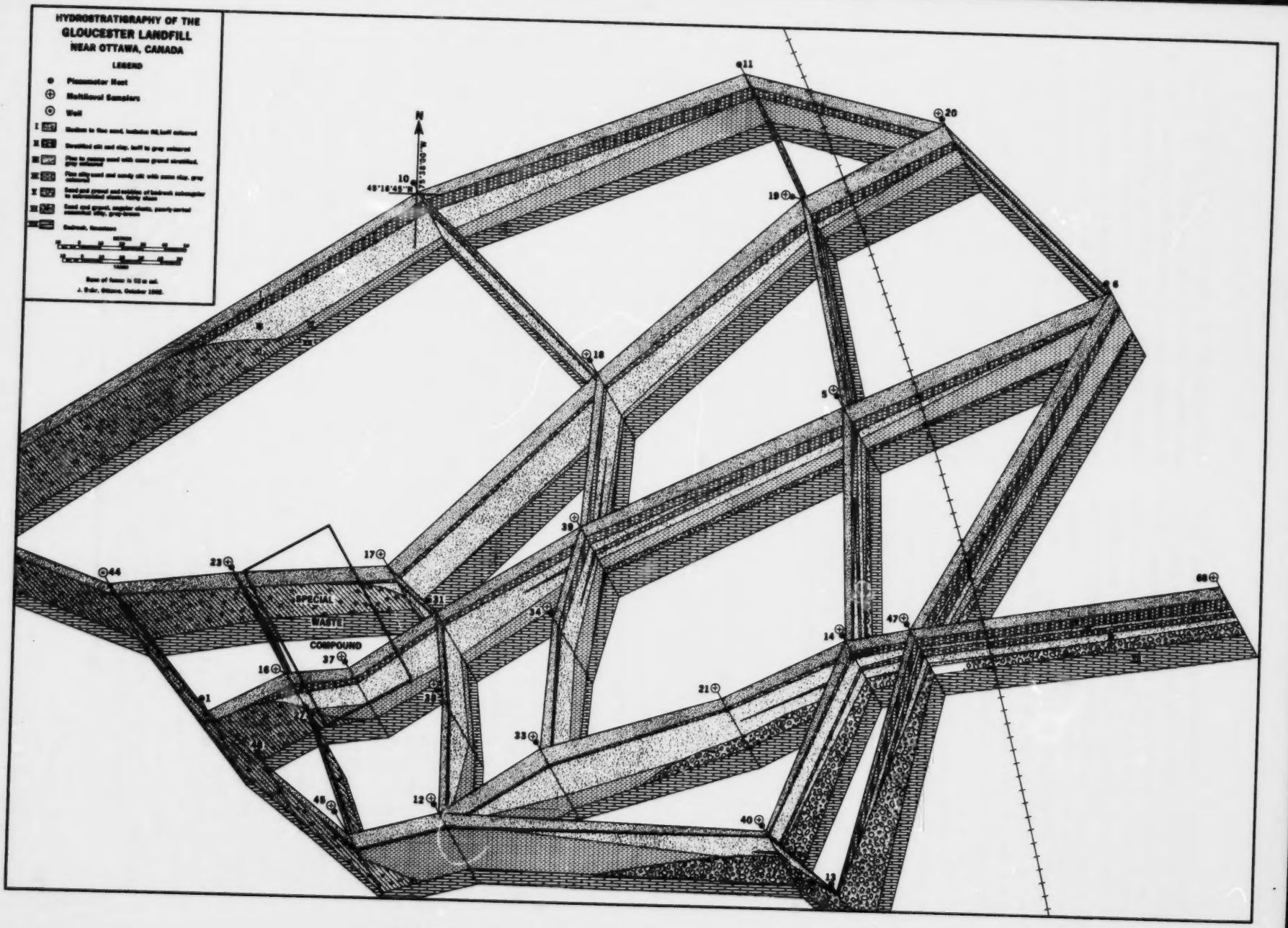


Figure 9. Fence diagram of Gloucester hydrostratigraphy by J. Bahr (GTC, 1983).

single multilevel device could be related to sediment type. Sampling points located in sand and gravel units yielded water at faster rates, for a given pump speed, than points seated in silt or clay zones. In fact, some points within silt or clay units yielded virtually no water when connected to a peristaltic pump operating at its maximum suction. At multilevel locations for which information on sediment type had not been collected during drilling, this correlation between production rate and sediment type was used to infer the hydrostratigraphy. Multilevel and piezometer locations are shown on Figure 10.

Thirteen additional sets of borings were made in November of 1982 and April of 1983 for multilevel and piezometer installation near the Special Waste Compound adjacent to the landfill and at the ends of Quinn Road and Delzotto Avenue. These holes were carefully logged to provide additional hydrostratigraphic information. The sediments recovered in split spoon samples for 67M, just south of the Special Waste Compound, confirmed the hydrostratigraphic interpretation in the vicinity of multilevels 37M, 38M, and 12M. The sediments encountered when drilling 68M and 68P at the end of Delzotto Avenue were consistent with the stratigraphy inferred in the vicinity of 14M and 47M. The fence diagram was extended to the west by inclusion of 68M. Monitor locations 69M, 70P and 71P, at the end of Quinn Road, were too far from the other monitoring sites to be included on the fence diagram, but again, the stratigraphy observed during drilling and split spoon sampling was consistent with a northeastward extension of the cross sections through holes 11P and 20M. Borings in locations 73P and 74P, to the north of the landfill, also encountered hydrostratigraphic units previously identified to the south.

The seven units, including the bedrock, were distinguished primarily on the basis of their hydraulic characteristics, that is, their capacity to store and transmit water, since these properties determine which units are potential conduits for contaminant migration. As such, a

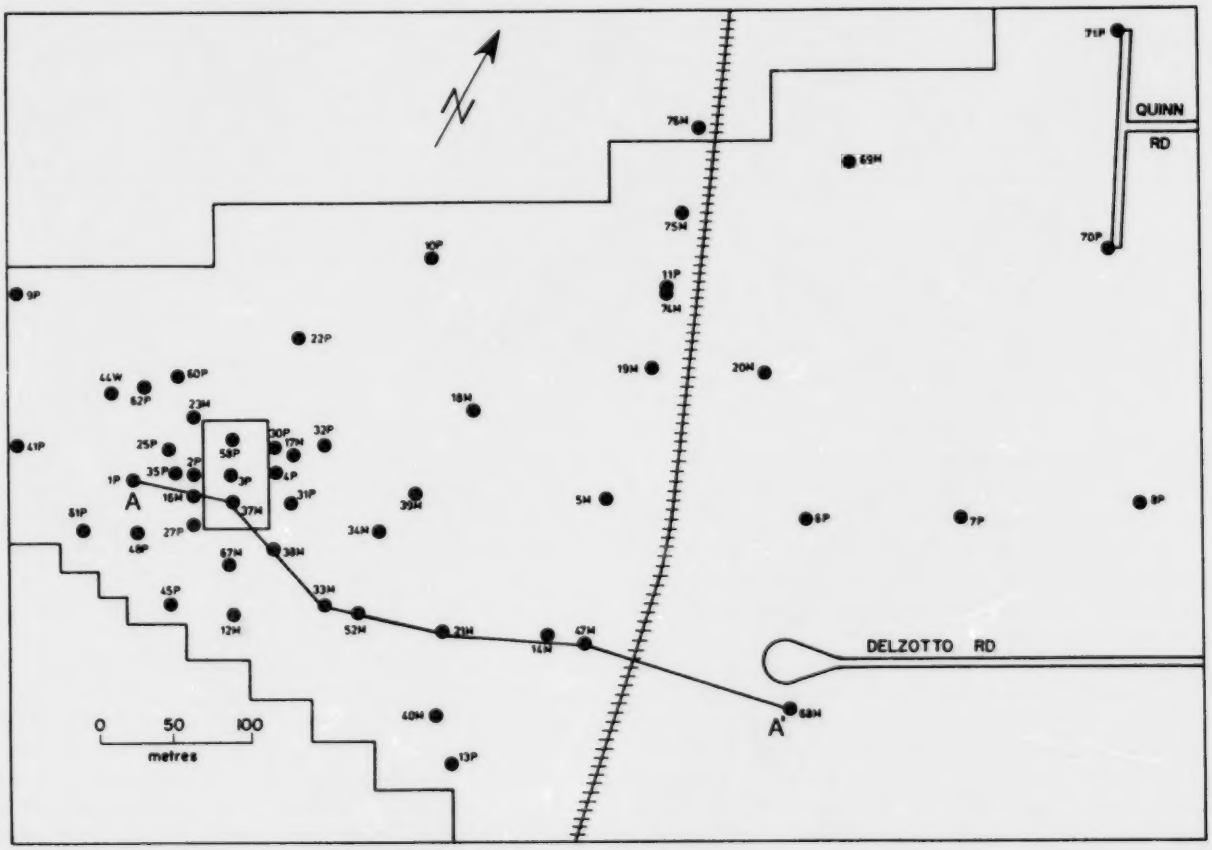


Figure 10. Locations of multilevel (M) and piezometer (P) installations.

single hydrostratigraphic unit may include several sedimentary units with distinctly different depositional origins. Nonetheless, the hydrostratigraphic units identified at the Gloucester site appear to form a coherent sequence of strata that can be characterized both on the basis of hydraulic properties and sedimentary nature.

Depositional Environment

The deposits in the vicinity of the Gloucester Landfill should be considered in the context of their presumed environment of deposition. One model of a generalized subaqueous outwash environment (after Reading, 1978) is depicted in Figure 11. Near the ice front is a succession ranging from unstratified to stratified till. Some fine sediment in fresh glacial meltwater rises with the freshwater owing to density differences between fresh and saline water, assuming the glacier terminates in the sea rather than in a freshwater lake. Silt and clay eventually settle out in the still water zone away from the glacier front. Sand and some coarser sediment may be transported as a high density underflow to some distance from the ice front, particularly at the mouths of channels at the base of the ice sheet. According to the models proposed by Rust and Romanelli (1975) and Cheel (1982), these channels generate subaqueous esker fans which form ridges oriented approximately in the direction of glacier retreat. Rust and Romanelli (1975) also postulate that multiple outwash channels depositing material at the ice front, coupled with intermittent glacier retreat, would give rise to a series of linear features parallel to the ice front. These deposits would be regarded as a type of morainal ridge. These authors proposed a stepped ice front model to explain the seemingly contradictory combination of asymmetric internal ridge facies, suggestive of a stationary ice front, with a ridge orientation which is more consistent with paleocurrents and the direction of glacial retreat. Three depositional models for subaqueous outwash ridges are illustrated in Figure 12 (after Rust and Romanelli, 1975).

Hydrostratigraphic Units

With these depositional models as a general framework, the seven units shown on Figure 9 were defined.

Unit VII

At the Gloucester Landfill, the basal hydrostratigraphic unit is the bedrock, an Ordovician limestone of the Oxford Formation. The private wells along Quinn Road and Delzotto Avenue derive their water from this formation. Judging from well logs compiled by the Ontario Ministry of the Environment, productive zones are encountered at depths of about 7 to 20 m below the bedrock surface and probably owe their high conductivities to the presence of fractures or solution cavities. One core sample of the bedrock was obtained from 14P-3. Examination of the core indicates that at this location, the limestone near the contact with unconsolidated sediments is gray, hard, and fresh, showing no evidence of intense fracturing or solution cavities. In addition to 14P-3, which was drilled into bedrock, a number of the borings completed by Gartner, Lee Associates (1978) are reported to have reached the bedrock surface. The recent borings at 67M, 68M, 69M, 70P1, 71P3, 72P1 and 78P2 also extend to bedrock as indicated by auger refusal at the bottoms of the holes. However, over much of the area of the fence diagram, bedrock topography could not be inferred from drilling records.

NHRI undertook a surface resistivity survey in the summer of 1982, with one objective being the identification of depth to bedrock beneath the Gloucester Special Waste site. The results of this survey indicate that depth to bedrock decreases rapidly to the east. The thickness of surficial deposits as estimated using resistivity data is only 8 m at locations about 750 m northeast of 68M. This value is consistent with well records which show bedrock at a depth of 6 m in the same area. Unfortunately, other results of the survey are less convincing. They suggest, in general, a very irregular bedrock topography beneath and to the south of the landfill. The known bedrock elevation at 14P-3 is 77.2 m above mean sea level, but resistivity measurements from a location about 60 m to the northwest indicate a bedrock elevation of only 52.9 m. This implies a bedrock surface dipping about 40°. This steep slope does not seem reasonable in an area where well logs indicate a relatively regular bedrock topography. These anomalous results cast some doubt on the usefulness of the resistivity method for bedrock prospecting at this site. Bedrock depths shown on the fence diagram should be considered approximate at best.

The limestone bedrock at depth may be considered to be a low permeability base to the upper weathered limestone and surficial sedimentary units. A fresh, unfractured limestone or dolomite generally has a hydraulic conductivity of less than 10^{-7} m/s (Freeze and Cherry, 1979), which is several orders of magnitude less than the estimated hydraulic conductivities of the unconsolidated aquifer units in contact with the bedrock at the Gloucester site. It may be expected that the upper portion of the limestone has been weathered, resulting in a more permeable zone with a hydraulic conductivity several orders of magnitude higher than that for intact limestone or dolomite.

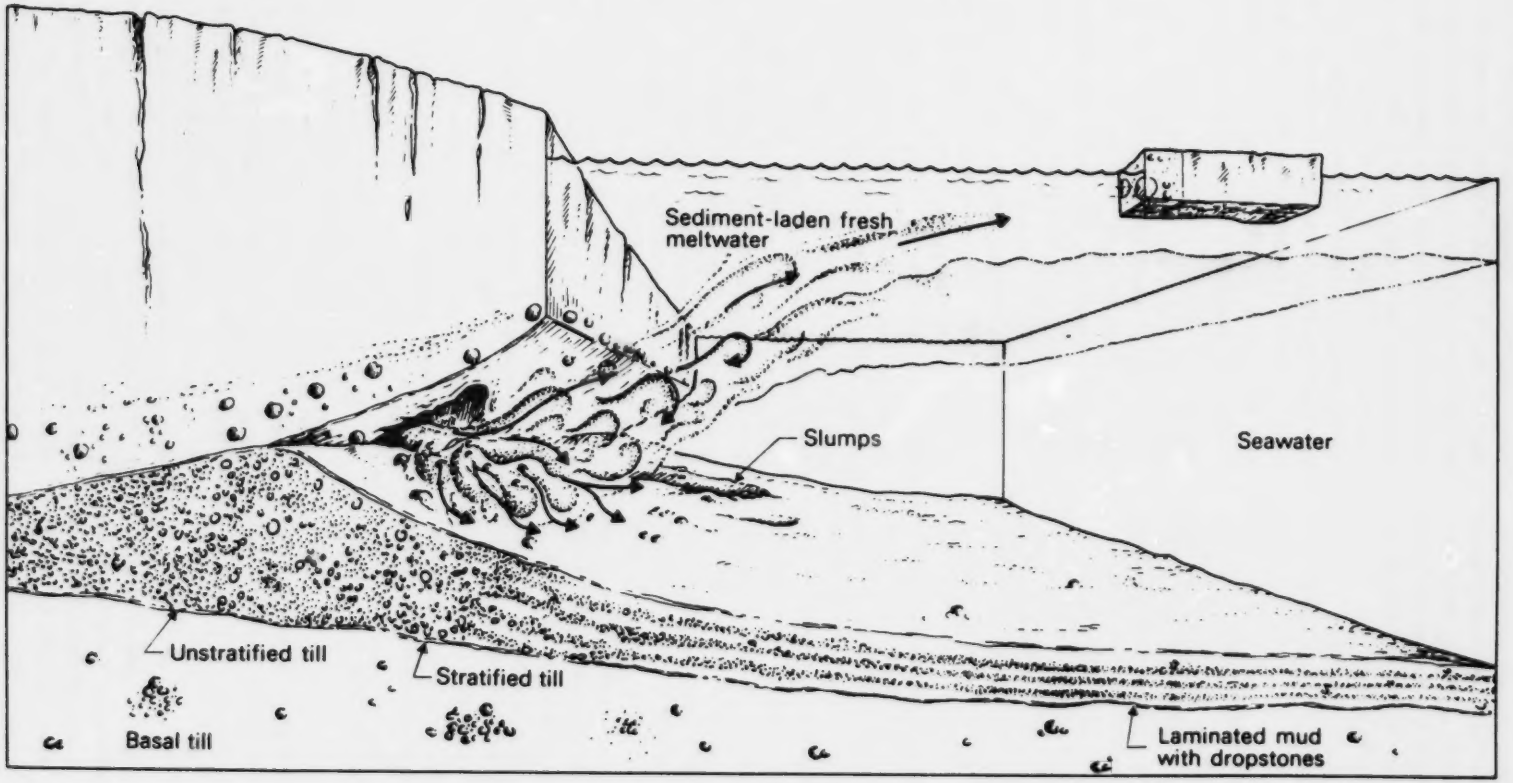


Figure 11. Hypothetical model of glaciomarine sedimentation (after Reading, 1978).

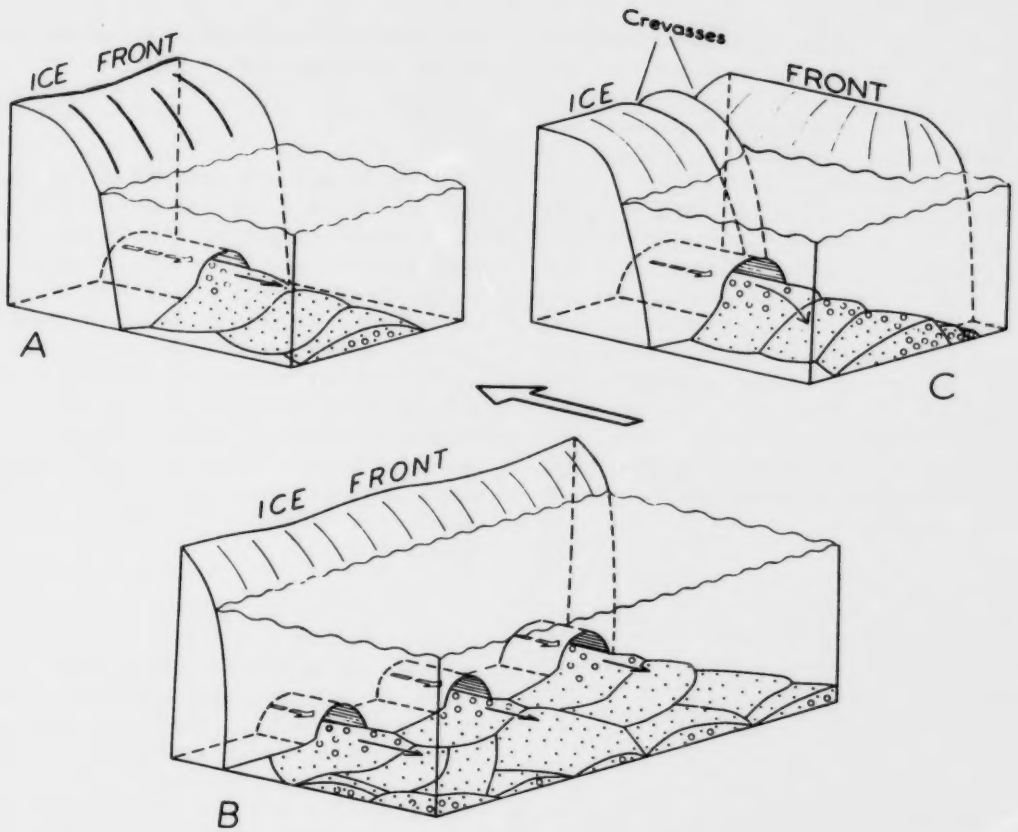


Figure 12. Three models for subaqueous outwash in ridges at a glacial ice front (from Rust and Romanelli, 1975).

Unit VI

Two coarse-grained sedimentary units overlie the bedrock. In the western portion of the area illustrated on the fence diagram is a unit composed of poorly sorted sand and gravel, designated unit VI. This unit is located at the edge of the surface expression of the outwash ridge and may be representative of the central facies of the ridge deposits. According to the records of French and Rust (1981), who completed one boring in the area underlain by this unit, clasts may exceed 3 cm in diameter. The log of well 26W and samples from piezometers 45P and 48P indicate that this unit contains silty fines. The poor sorting of this unit may indicate deposition near the ice front either as a proximal unit in an esker fan or as a morainal deposit at a stationary ice front. A representative grain-size curve is shown in Figure 13.

The hydraulic conductivity of this unit would be expected to be somewhat lower than that of a clean sand or gravel due to the presence of silty fines. Hydraulic conductivities calculated from slug tests in piezometers 48P-1 and 48P-2 range from 2.9×10^{-5} to 1×10^{-4} m/s.

Unit VI terminates abruptly in the vicinity of the Special Waste Compound. It is also shown terminating in a steep, eastward dipping contact with units IV and III to the north and south of the compound. A more accurate representation of this contact would probably be a gradational transition from poorly stratified to stratified deposits laid down at a greater distance from the ice front and/or the ridge axis.

Unit V

The second coarse-grained unit in contact with the bedrock occurs in the southeastern portion of the fence diagram. This unit consists of medium to coarse sand and gravel with some cobbles (see grain-size distribution, Fig. 13). Samples from the bases of piezometers 40P and 47P contain very few fines, indicating that unit V is better sorted than unit VI. Samples from a boring near 39M

contain abundant clasts of limestone in the sediments recovered from near the presumed bedrock contact, suggesting a local source for at least some of the deposit. The unit thins rapidly to the east from a thickness of over 10 m in 40M to less than 1 m in holes 7 and 8 logged by Gartner, Lee Associates (1978) to the north of Delzotto Avenue. However, it extends north at least as far as the oval horse track (Fig. 4) west of Quinn Road, since 2 m of gravel and cobbles was encountered at the base of hole 69M. In this location, silty and clayey fines were recovered along with the gravel. These fines may have come from the gravel unit or they may be from the thick, fine-grained unit above. In its northernmost and westernmost extremes, this unit may be simply a thin mantle of bedrock debris. However, it seems plausible that in its thickest zone, unit V constitutes an older outwash ridge that was reworked to some extent following glacial retreat before being buried by finer grained sediments.

Four slug tests in this unit were conducted in 47P-1 and indicate an average computed hydraulic conductivity of 8.1×10^{-5} m/s. Analysis of two long-term pumping tests conducted on well 49W resulted in computed hydraulic conductivities ranging from 1.3×10^{-4} to 1.2×10^{-3} m/s.

Unit IV

Overlying portions of the bedrock and overlying or adjacent to portions of units V and VI is a dense, fine-grained, silty sand and sandy silt containing some thin beds of clay. Split spoon samples of this unit from borings 67M and 69M indicate horizontal stratification or a fine massive structure. In 67M a 0.5 m thick, fining upward sequence of gravel to fine sand was encountered in the upper portion of this unit. In the context of the glaciomarine sedimentation model presented in Figure 11, this unit appears to represent a fine-grained portion of the stratified facies deposited in moderately deep water at an intermediate distance from the ice front. A subaqueous esker fan extending away from the ice sheet would be expected to have fine-grained distal facies generated by a combination of settling of silt- and clay-sized particles from the water column and low-energy bottom transport of fine outwash sediment.

As a hydrostratigraphic unit, these fine sediments function as a confining layer over a large area of unit V and portions of unit VI. They also separate unit III from the underlying bedrock. Unit IV ranges in thickness from a few metres in the vicinity of 18M to over 10 m to the south and on the east side of the railroad tracks. A slug test conducted in piezometer 45P-2, which terminates in this unit, yielded a computed hydraulic conductivity of 5×10^{-6} m/s. Multilevel samplers 12M and 45M do not produce water from sampling points located in this unit.

Unit III

Unit III consists of stratified sand and gravel beds exhibiting upward fining sequences in some areas. The matrix ranges from silty to clayey in some beds, while others contain few fines; see for example, the grain-size distribution curve in Figure 13. Unit III probably grades into unit VI to the west. To the east, it interfingers with the silts and clays of unit II. As a distal facies of a subaqueous esker fan, the sediments of unit III may have been deposited episodically during phases of high outflow from the channel beneath the ice mass. The amount of fines in the matrix could represent the degree of mixing with material settling from the water column and hence could be an indicator of the rate of transport and deposition of the coarser material.

This unit reaches its maximum thickness of about 16 m in the vicinity of borings 33 and 31, east and southeast of the Special Waste Compound. Farther east, it thins rapidly and near the railroad tracks interfingers with the finer grained units II and IV. Hydraulically, it is connected to unit V in areas where the lower confining unit is absent. It is also hydraulically connected to unit VI at its westernmost extent. A series of pump tests on well 49M produced responses in this unit, indicating a mean hydraulic conductivity of 3.1×10^{-4} m/s. Slug tests in this unit yielded hydraulic conductivities on the order of 10^{-5} to 10^{-4} m/s. The considerable range in hydraulic conductivity with depth inferred from slug tests, as well as the complex patterns of contaminant migration inferred from measured concentrations of organic compounds in water from multilevel samplers, suggests significant variation in this unit.

Unit II

Unit II acts as the principal confining layer for most of the lower aquifers. It is composed of silts and clays and is the least permeable of the unconsolidated deposits. As it covers both units III through V and portions of unit VI, a reasonable origin for this unit would be slow settlement from the water column in a marine or lacustrine environment. This deposition occurred after the ice sheet had retreated from the position it occupied during formation of the outwash ridge.

Unit II is absent in the western area of the site, perhaps removed by wave reworking during the retreat of the Champlain Sea. It is also absent in the vicinity of 13P,

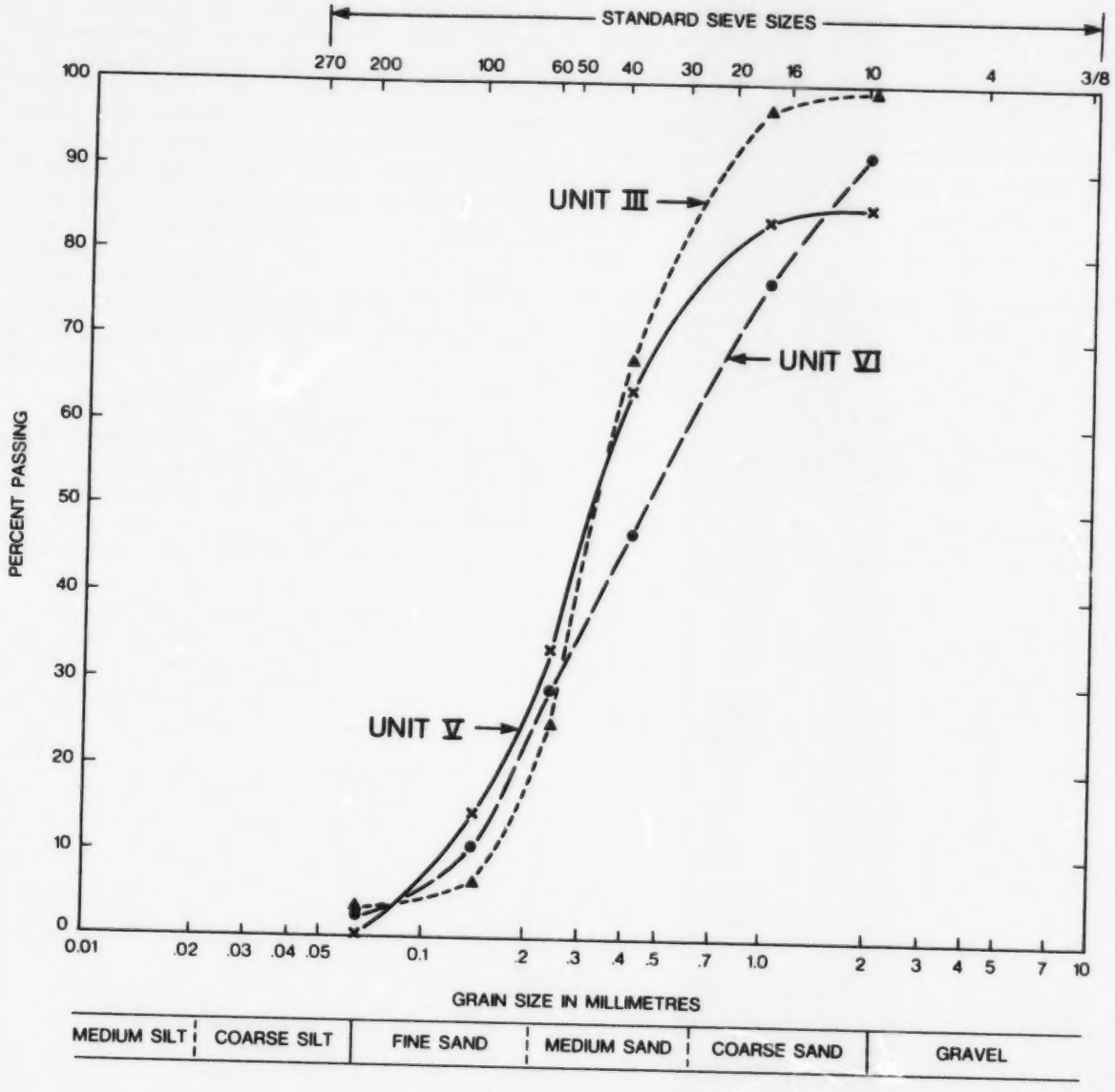


Figure 13. Grain size curves from units III, V and VI.

in the area of maximum thickness of unit V. In areas where this upper confining unit is missing, water levels in piezometers do not vary with depth as they do over most of the area. Unit II may also have been breached in some portions of the landfill when trenching extended below the water table. The thickness of the silts and clays is estimated as less than a metre near multilevel 33M but may be nearly 5 m in the vicinity of multilevels 39M and 5M. Multilevel sampler points installed in this unit have very low production rates, and if they do produce, the water is silty.

Unit I

The surface deposits over the entire area are designated as unit I on the fence diagram. This unit includes not only the fine to coarse sand that makes up the most recent sediments at the site but also several metres of landfill material, a mixture of garbage and soil. Marine fossils were found in an irregular block of sandy silt exposed in a trench in the upper sediments excavated near 48P. Rust (1977) describes similar fossiliferous blocks in an exposure in the Stittsville Ridge, another subaqueous

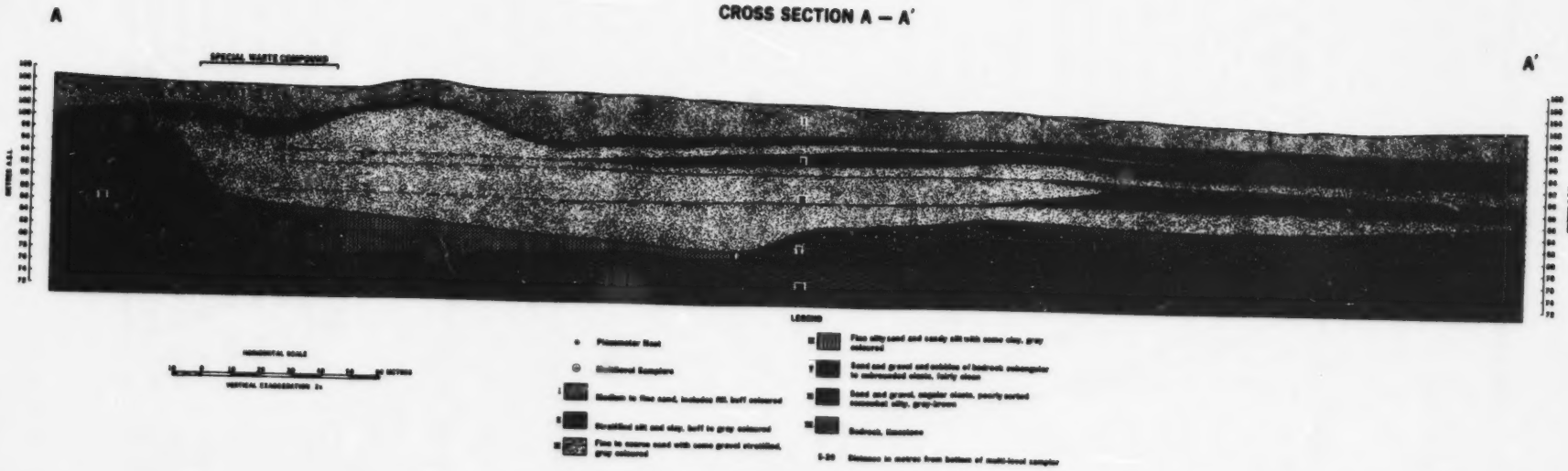


Figure 14. A–A′ cross section from GTC hydrostratigraphic model.

outwash deposit in the Ottawa area. He ascribes these blocks and the surrounding contorted beds of fine sand to deformation during glacial overriding during a late readvance of the ice sheet into the Champlain Sea. Gadd (1978) argues that deformation of the fossiliferous strata in the South Gloucester area cannot be attributed to the presence of active glacial ice but must be related to environments in the shoaling phase of the Champlain Sea. Whatever the nature of the deformation structure, the presence of these shells in unit I indicates deposition in a marine environment. Subsequent reworking by geologic processes and landfill or agricultural activity may have served to modify the structure and permeability of this unit.

The upper sediments are saturated below a depth of several metres and form a water table aquifer. This aquifer is contaminated as a result of both municipal waste disposal and burial of chemicals in the Special Waste Compound. Two pump tests were attempted on well 28W which is screened in this unit. The results indicated a very low hydraulic conductivity, on the order of 5×10^{-7} m/s. Results of a slug test in 30P-6, which appears to be in the base of unit I, indicate a slightly greater conductivity of 6.1×10^{-6} m/s. Even this conductivity is almost two orders of magnitude lower than that indicated for the other aquifer units, implying that transport of contaminants may be much slower in the water table aquifer than in the confined zone.

A–A′ Cross Section

Figure 14 shows one fence from the GTC hydrostratigraphic model – in this case the cross section along which contamination from the Special Waste Compound has apparently migrated.

THE NHRI HYDROSTRATIGRAPHIC REINTERPRETATION

Four multilevel samplers, two deep piezometers and two bedrock wells were installed at the site during the

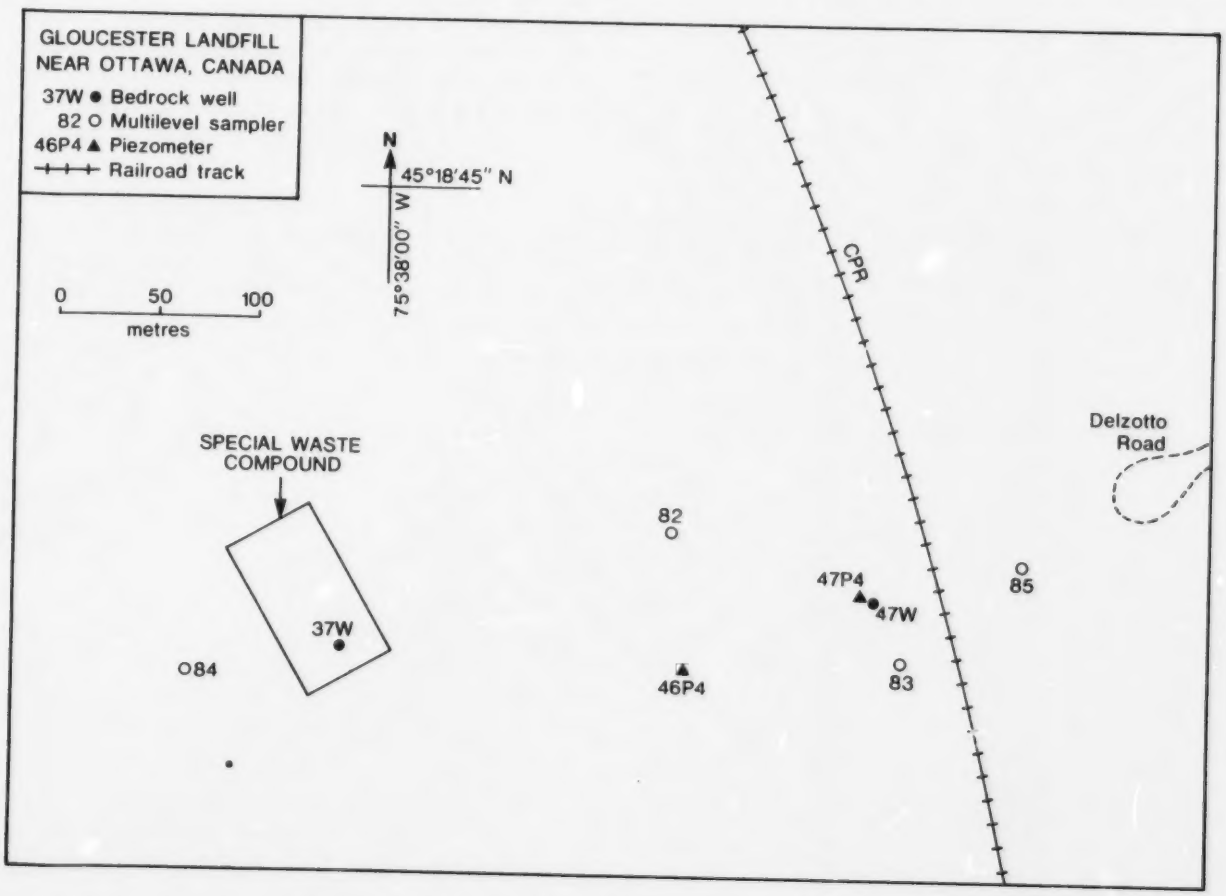


Figure 15. Locations of hydrogeologic instrumentation installed in 1983.

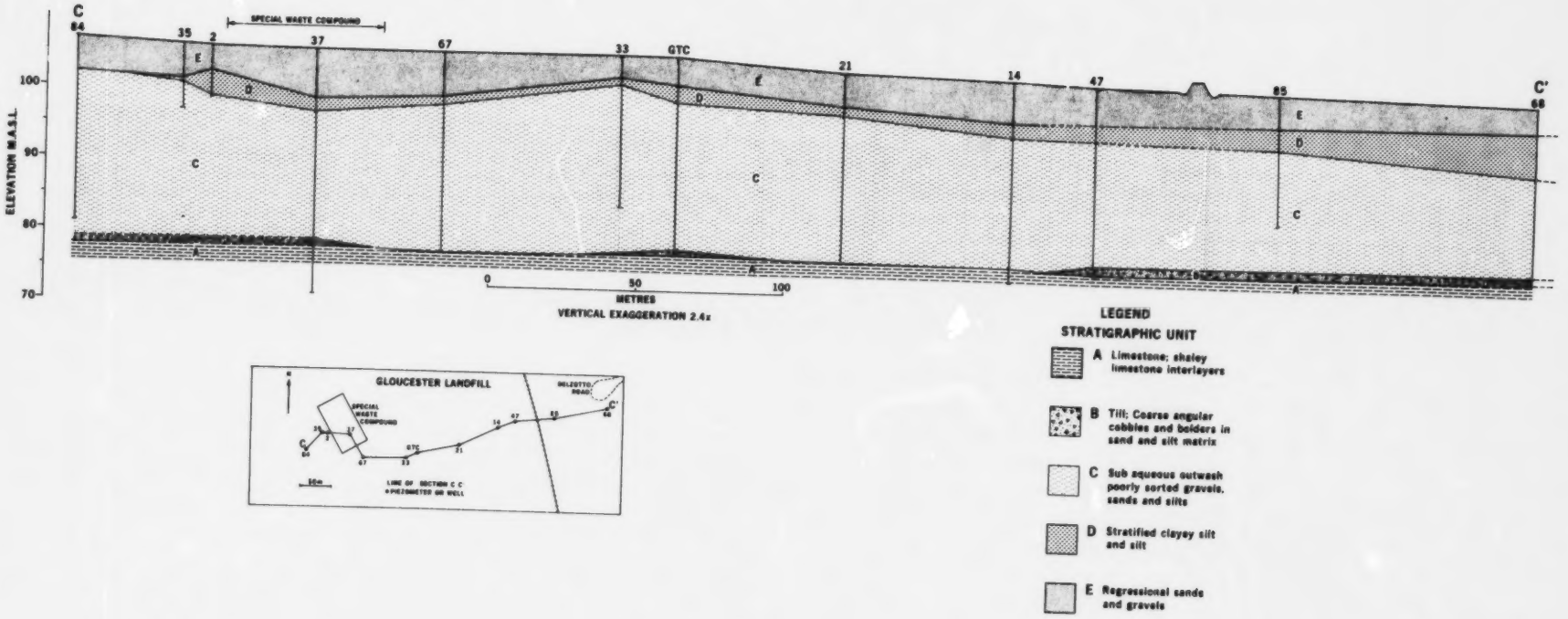


Figure 16. C-C′ cross section of NHRI.

summer of 1983 (locations shown on Fig. 15). At the multilevel locations, numerous split spoon and core samples of the sediments were collected and analyzed (logs are shown in Appendix A).

Examination of the sediments and cores, evaluation of the conditions encountered during drilling, and consultation with N. Catto, University of Alberta, indicated that revisions should be made in the hydrostratigraphic interpretation for the site. The cross section in Figure 16 illustrates the new interpretation which includes five designated units.

Unit A, the bedrock, is encountered at depths between about 25 and 30 m. Examination of cuttings recovered during the drilling of the bedrock wells indicates that the rock is predominantly gray limestone (dolomite?) with interstratified dark gray, shaley limestone. Open fractures and cavities are present within at least the upper 7 m of bedrock.

At several locations, up to 2 m of dense, coarse material (unit B) was encountered above the bedrock contact. This discontinuous unit is interpreted as a basal till. Pumping of piezometers screened in this unit indicates a relatively low permeability consistent with poor sorting that is characteristic of tills.

Unit C, overlying the till or bedrock, is a thick (up to 25 m) interstratified and interfingering sequence of silts, sands and poorly sorted gravels. This complex unit is believed to have been deposited as coalescing and overlapping subaqueous esker fans that formed beneath standing water adjacent to the glacial ice front (Rust and Romanelli, 1975). The principal axis of sediment discharge was likely along the regional northwest-southeast trending ridge which flanks the site on the south and west. The gravels are generally very poorly sorted, reflecting rapid deposition and, in some cases, probably mass flow. Finer-grained strata in the sequence represent the more distal facies of the fans or the final depositional phases of periodic sediment influxes. The presence of graded sequences in cores from 82M provides evidence that deposition was episodic. Due to the complexity of the sequence, further subdivision or correlation of individual strata is not practical. In general, however, the proportion of gravel decreases in the upper part of the unit and to the north and east away from the northwest-southeast trending ridge.

The transition between units C and D is marked by a gradual decrease in the proportion of coarse layers. In unit D, most of the strata are clayey silts and silts that were deposited after the ice front had retreated from the area but the sea was still relatively deep. The scattered occurrence of large boulders in the fine materials is likely the result of ice rafting.

Unit E, which may be up to 10 m in thickness, represents the regressional facies of the Champlain Sea. Sediments range from fine sands to gravels, but in general, the coarser materials predominate in the south and west parts of the area near the northwest-southeast trending ridge. The coarse gravels near the ridge appear to be lag deposits formed as a result of erosional reworking of the underlying strata near the shore of the sea. The pinch-out of unit D in the western part of the site is probably a result of erosion during the regression of the shoreline. The distribution of this unit near the pinch-out is likely very irregular.

The Ground-Water Flow System

HYDROGEOLOGICAL PROPERTIES OF THE AQUIFER MATERIALS

Surficial Sediments

The principal hydraulic parameters – hydraulic conductivity (K), transmissivity (T) and storativity (S) – describing the ground-water flow system were determined by a series of single piezometer (i.e. slug) tests and short- and long-term pump tests.

Single piezometer tests were used to determine *in situ* hydraulic conductivities for the porous media in the vicinity of the piezometer. The test consists of removing (or adding) a known volume of water from (or to) the piezometer and observing the recovery of the water level with time. For a single point piezometer, the method of Hvorslev (1951) provides the simplest interpretation of water level recovery data. This test procedure was used to determine hydraulic conductivities in all of the hydrostratigraphic units at the site. Table 8 provides a list of average hydraulic conductivities for each unit determined on the basis of 100 tests on 33 piezometers. Hydraulic conductivities calculated for each test on each piezometer can be found in Appendix B.

Table 8. Results of Single Piezometer Response Tests

Stratigraphic unit	No. of piezometers	No. of tests	Average K (m/s)
E	4	11	4.1×10^{-5}
D	5	9	7.9×10^{-6}
C	24	80	1.1×10^{-4}

The data in Table 8 indicate that the outwash aquifer (unit C) has an average hydraulic conductivity of about 1.1×10^{-4} m/s which corresponds to clean sand. This unit is stratigraphically and sedimentologically complex and consequently shows a wide range of values. In general, the response tests for unit C indicate an increase in hydraulic conductivity with depth probably as a result of an upward decrease in the proportion of the coarser outwash facies.

Unit D has an average hydraulic conductivity of approximately 8×10^{-6} m/s, which is typical of a silty sand or silt. This value is more than an order of magnitude lower than that for unit C, and therefore unit D should be an effective confining layer or aquitard. Unit E, the surficial regressional sequence, contains a water table aquifer and has an hydraulic conductivity of about 4×10^{-5} m/s which is consistent with a silty sand.

Pump tests are specifically suited to determining average hydraulic conductivities, transmissivities and storativities for relatively large volumes of the aquifer. The methodology consists of pumping a well for a period of time and monitoring the water levels in a number of adjacent piezometers. As the water level drawdown increases due to pumping, piezometers at greater and greater distances are affected.

At the Gloucester site, a number of short- and long-term pumping tests were conducted during the summer of 1982. Short-term tests (1-2 h) were used to determine the local hydraulic properties (K, T, and S) of the aquifer, and long-term tests (8-12 h) were used for the same purpose and also to delineate boundary effects and evaluate the continuity of the confining layer (unit D).

All of the pump tests conducted in the outwash aquifer were centred at well 49W, which is screened over the depth interval 14.9 m to 21 m. For the long-term pump tests, up to 36 piezometers were monitored intermittently, and for the short-term tests, only the few adjacent piezometers were measured, although the monitoring interval

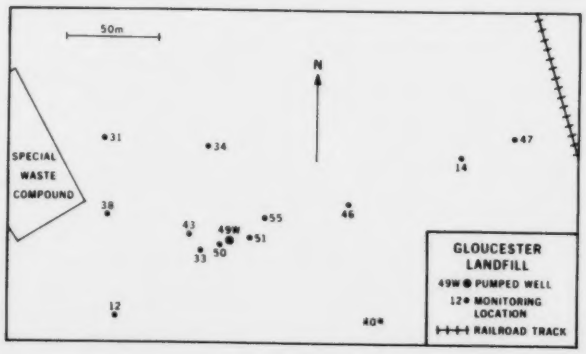


Figure 17. Plan view of southern part of the Landfill showing the location of monitors used during the series of pump tests.

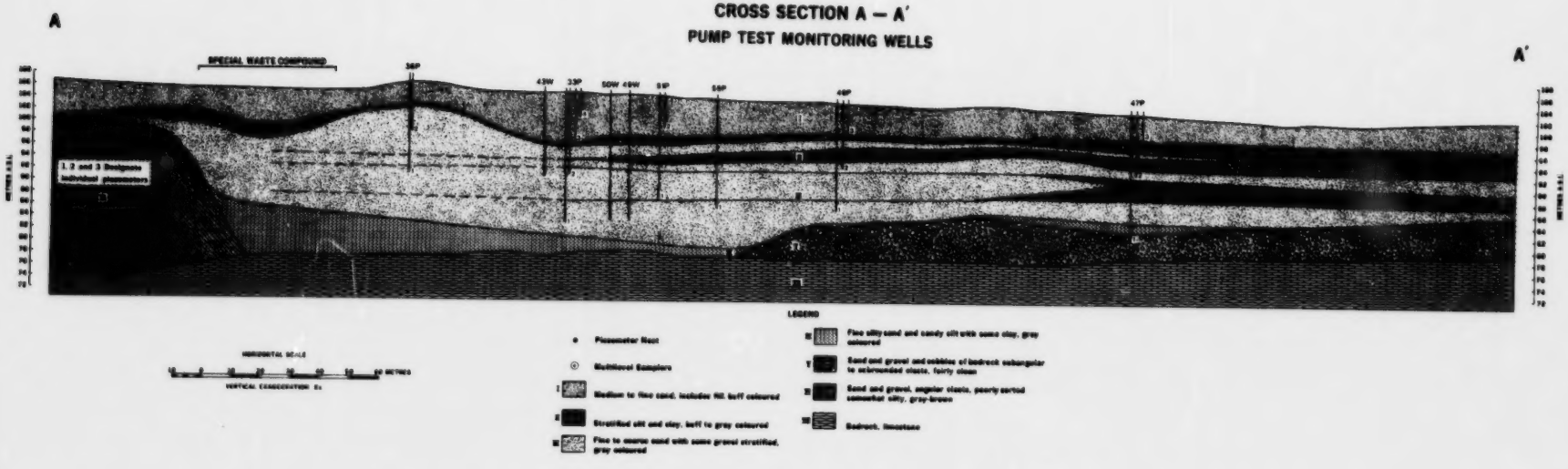


Figure 18. Cross-sectional view showing the location of the monitors used during the pump tests.

Table 9. Results of Short- and Long-Term Pump Tests on Well 49W

Stratigraphic unit	Range of transmissivity (m^2/s)	Mean transmissivity (m^2/s)	Mean storativity	Mean hydraulic conductivity (m/s)	Range of hydraulic conductivity (m/s)
C	1.2×10^{-3}–3.4×10^{-2}	9.8×10^{-3}	2.5×10^{-3}	4.9×10^{-4}	6.0×10^{-5}–1.7×10^{-3}

was more frequent. Monitored piezometer locations are shown in Figure 17 in plan view and in Figure 18 in cross section. The discharge rate for the pump tests ranged from 1.8×10^{-3} to 2.3×10^{-3} m^3/s (23-30 gpm). The discharged water was directed into a pit containing sorbent materials in the northeast corner of the Special Waste Compound, a distance of 150 m from 49W.

Drawdown data for each piezometer monitored during the pump test were analyzed by two methods: log-log type-curve matching (i.e. Theis method) and the semi-log plot, or Jacob method. Descriptions of both methods can be found in Freeze and Cherry (1979).

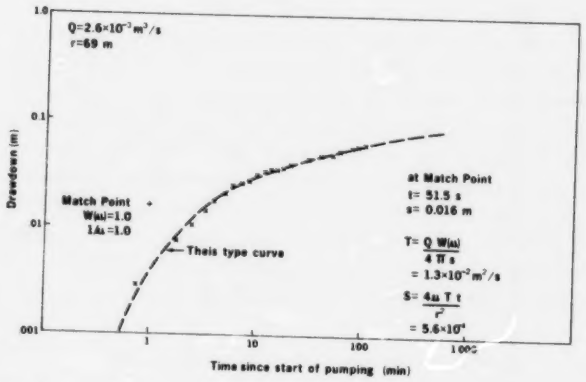


Figure 19. Theis curve analysis of time vs. drawdown at piezometer 46P-1.

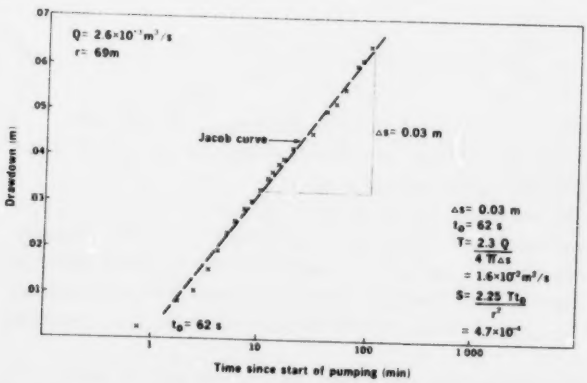


Figure 20. Jacob curve analysis of time vs. drawdown at piezometer 46P-1.

Examples of the Theis and Jacob methods for analyzing the data from piezometer 46P-1 are shown in Figures 19 and 20, respectively. In cases where the data were a poor fit to the Theis curve, values of transmissivity and storativity were not calculated.

Table 9 presents average values of aquifer transmissivity, storativity, and hydraulic conductivity for unit C calculated from the results of monitoring 23 piezometers during the pump tests. Hydraulic conductivities were determined from the transmissivity by dividing by an estimate of the average unit C thickness. Calculated values of T, S, and K for each piezometer can be found in Appendix C.

The high values of transmissivity and storativity show the outwash aquifer to be a high yield, very productive unit. Calculated hydraulic conductivities for unit C (6.0×10^{-5} to 1.7×10^{-3}) compare favourably with those calculated from the single piezometer test results. Pump test values for some of the lower parts of unit C are higher than the slug test values, probably due to presence of nearby gradational boundaries causing hydraulic conductivity values to be representative of the mean of several strata.

In addition to the time-drawdown methods for analyzing data from each piezometer, transmissivity, storativity and hydraulic conductivity can also be calculated from distance-drawdown curves. This method is illustrated in Figure 21 for a long-term pump test at well 49W using

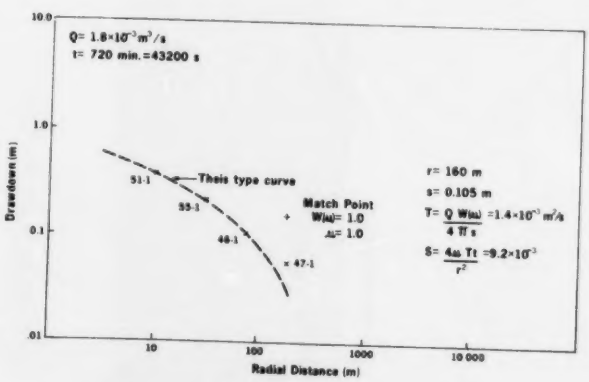


Figure 21. Theis curve analysis of distance vs. drawdown along C-C' cross section.

drawdown results for piezometers 51P, 55P, 46P and 47P at radial distances of 10, 30, 70, and 165 m, respectively. All of these piezometers are screened in unit C and yield average values of transmissivity of 2.2×10^{-3} m^2/s, storativity of 8.6×10^{-3} and hydraulic conductivity of 1.6×10^{-4} m/s. These values are somewhat lower than those determined from individual piezometers (Table 8).

Long-term pump tests were also used to confirm aspects of the hydrostratigraphy, especially to discern boundary effects and to evaluate the continuity of unit D. Increases in the rate of drawdown after about 6 h of pumping were noted in most observation piezometers. These results suggest that the expanding cone of depression was intersecting lower permeability boundaries. The relatively pronounced effect in piezometers 12P-1 and 40P-2 is consistent with stratigraphic data, which indicate that the outwash sequence (unit C) is characterized by a much higher than average proportion of finer facies at these locations. Increased drawdown in piezometers such as 47P-1 and 47P-2 probably reflects the eastward and northward fining trend in unit C.

The pump tests were also used to examine the continuity and confining capacity of unit D. All shallow water table piezometers monitored to the north and east of 49W did not respond even after 12 h of pumping. In this area, unit D appears to be continuous and adequately confines the lower aquifer. To the south and west this is not likely the case. The shallowest piezometers at nests 33P and 40P responded immediately to pumping. Although these piezometers may penetrate the silt layer, it is thought that unit D becomes thin or discontinuous in this area.

Porosity of the aquifer materials was determined by Gartner, Lee Associates Ltd. (1978) for all stratigraphic units and by NHRI for unit C. The outwash aquifer has a porosity of 0.30–0.35, while the upper aquifer has a range of 0.35–0.40. The confining layer unit D has a porosity of about 0.40–0.45.

Bedrock

The wells 37P-3 and 47P-5 (locations 37W and 47W in Fig. 15, respectively) were drilled by a mud-rotary rig into the limestone bedrock. A 10-cm (6-in.) casing was cemented approximately 3 m into the bedrock (Fig. 22). The holes were then re-entered and extended another 3.5 m. During the drilling, productive zones (i.e. loss of mud circulation interpreted as fractures or joints) were intersected in both holes. In 37P-3, fracture(s) were encountered near the bottom of the completed hole, whereas in 47P-5 the productive zone was at a shallow depth into the bedrock and was subsequently sealed with cement prior to advancing the hole. Consequently, 37P-3 and 47P-5 differ dramatically in terms of hydraulic characteristics; 37P-3 intersects a productive fracture zone and has a high well yield, whereas 47P-5 is likely completed predominantly in intact rock (closed or tight fractures) and has a low well yield. As a result of this contrasting nature, it was necessary to apply different testing techniques in each hole. Piezometer 37P-3 was hydraulically stressed by pumping at 270 L/min and the drawdown response in surrounding piezometers monitored for a period of over 3 h. In piezometer 47P-5, sustained pumping was not feasible because of the low well yield. Instead, an overburden piezometer, 47P-1, directly above 47P-5 was pumped at about 22 L/min for 3.5 h and the response of the bedrock piezometer monitored. In addition to the pump test, 47P-5 was also slug-tested by quickly removing a large volume of water from the hole and monitoring the water level recovery.

Results of the pump test on 37P-3 are shown in Figure 22. Pumping at 270 L/min produced approximately 2 m of drawdown in 37P-3. Most of the drawdown (1.8 m) occurred in the first 3 min of pumping with only a 20-cm decline occurring during the rest of the test. Noticeable drawdown (up to 3 cm) was recorded in piezometers 37P-1, 3P-1, 4P-1, 30P-1, 2P-1, 38P-1 and 33P-1, indicating a radius of influence of up to 90 m. Upon stopping the pump, 38P-3 recovered immediately, while the surrounding piezometers showed a much slower recovery rate.

Analytical solutions to pump tests such as those described are difficult to obtain for the situations shown in Figure 22. In the case of 37P-3, only a small thickness (3 m) of bedrock is being tested, and this is likely connected to the overburden via fractures or joints. When pumping 37P-3 most of the water is probably derived from the overburden and flows downward into bedrock. Similarly, strong vertical gradients occur when 47P-1 is pumped because the piezometer is essentially a point source and water will flow to the piezometer from above and below. It is essentially due to vertical flow or partial penetration effects that typical well hydraulic methods cannot be applied to the cases described. It is possible, however, to make qualitative statements regarding the hydraulic connectivity between the bedrock and overburden.

Piezometer 37P-3 is well connected to the overburden probably through subvertical fractures or bedding joints and high-angle fracture sets. The fracture set is probably quite open, but its lateral extent is unknown. The large radius of influence when pumping indicates that either the lateral extent is large or the overburden above bedrock is permeable enough to transmit a head change over a large area. A preferred direction for drawdown might indicate a general orientation on the fracture set, but this is not

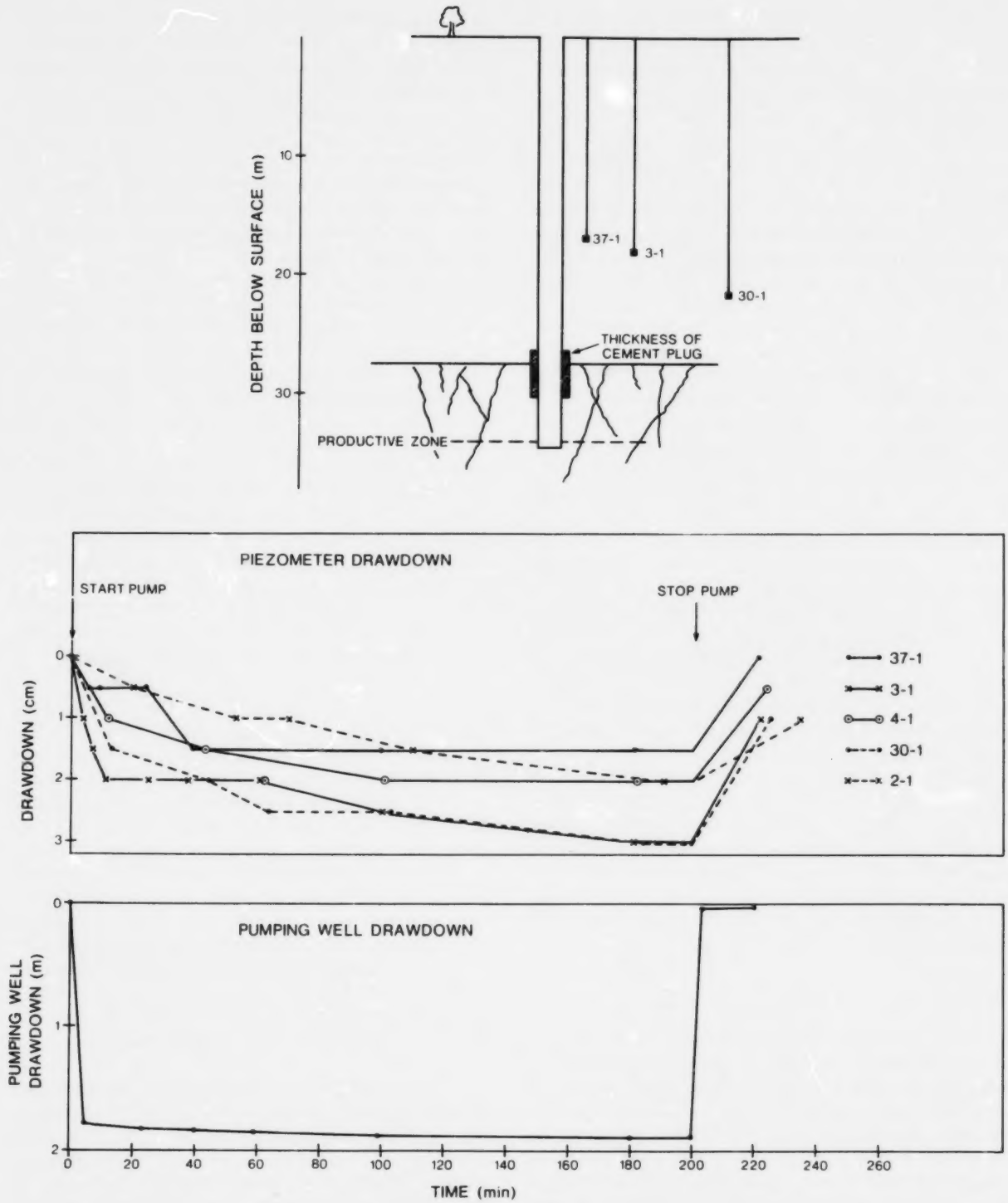


Figure 22. Hydraulic response of piezometers in the overlying outwash aquifer to the pumping of the bedrock well 37P-3. Inset shows locations of the piezometer screens.

indicated from the drawdown data. Thus 37P-3 appears to be hydraulically connected to the surficial aquifer and is susceptible to contamination from the overlying Special Waste Compound. Water level data at location 37P indicate that there is a downward gradient into the bedrock. Consequently, it is possible that 37P-3 could become contaminated in the future.

Piezometer 47P-5 also appears to be hydraulically connected to the overburden, but the nature of the connection is difficult to interpret. Pumping of 47P-1 causes a drawdown in 47P-5 and a more significant drawdown immediately above the bedrock. Slug test data indicate at least two different hydraulic conductivities exist in the one hole. One hydraulic conductivity would be that of the intact limestone bedrock. The other could result from a poorly cemented casing, increased permeability at some distance from the wellbore, a poorly cemented productive zone or a closed fracture in the intact rock. The response of 47P-5 to overburden pumping could mean that the casing is improperly set into the bedrock, but a poorly cemented productive zone may yield the same effect. Hydraulic head measurements indicate that there is a slight upward gradient between the bedrock and the overlying sediments.

In summary, piezometers 37P-3 and 47P-5 are both hydraulically connected to the surficial aquifer. Piezometer 37P-3 shows a significant connection probably through fractures and joints, while 47P-5 is connected to the overburden presumably by a leaky casing or a poorly cemented fracture.

GROUND-WATER FLOW PATTERNS

Throughout most of the eastern and northern parts of the Gloucester site, relatively large head differences occur across unit D, the silt and clayey silt stratum. These differences indicate that this unit behaves as an aquitard, separating a shallow water table aquifer from a deeper flow system in the outwash sequence (unit C).

Figure 23 illustrates the topography of the water table surface, and where unit D occurs, the configuration of the potentiometric surface for the deeper confined system. The location of the western and southern boundaries of the aquitard (unit D) has been inferred on the basis of stratigraphic and hydrologic information.

The pattern of water table contours indicates that shallow ground-water flow in the vicinity of the Special Waste Compound is generally directed toward the margin of the aquitard where downward movement occurs into the outwash sediments. Near the south part of the compound, the configuration of the contours and an apparent decrease in, or absence of, vertical gradients suggest that the aquitard is locally discontinuous and that downward flow occurs where it is absent. The holes in the aquitard may be natural or the result of excavating that accompanied waste disposal in this area. Aerial photographs (Figs. 3 and 4) taken during the operation of the site show that trenching was extensive adjacent to the south end of the Special Waste Compound.

The orientation of potentiometric contours suggests that ground-water flow directions in the deeper confined aquifer near the Special Waste Compound vary locally and may range from slightly south of east to northeast.

Over most of the area where both the water table and confined aquifers are present, vertical gradients across the aquitard are directed downward. In general, the magnitude of the vertical gradient is the greatest in the west and decreases toward zero to the east.

The distribution of hydraulic head in the lower confined aquifer along an approximate east-west section is illustrated in Figure 24 for three periods of measurement: October 1982, July 1983, October 1983. It is apparent from the configurations of the contours that seasonal and year to year changes occur in the flow pattern. These changes reflect variations in antecedent precipitation conditions, and for October 1983 near the Special Waste Compound, probably the withdrawals and injections associated with the GTC Purge Well Evaluation Project. In general, the data indicate that groundwater flow in the confined aquifer is horizontal toward the east. Although vertical components of flow are evident for each time period, the patterns are not consistent through time.

GROUND-WATER VELOCITIES

In situ measurements of ground-water velocity have been made in several piezometers at the Gloucester site using a modified point dilution apparatus. Results obtained by this technique provide a direct indication of the range of velocities in the flow system and serve as a basis for evaluating average velocities calculated indirectly using pump and slug test data and from the migration pattern of "bomb tritium," i.e. ^{3}H released during atmospheric testing of nuclear weapons in the 1950s and 1960s.

Point Dilution

The point dilution method relates the dilution of a tracer introduced into a well screen to the ground-water velocity in the aquifer near the well. The dilution rate of a

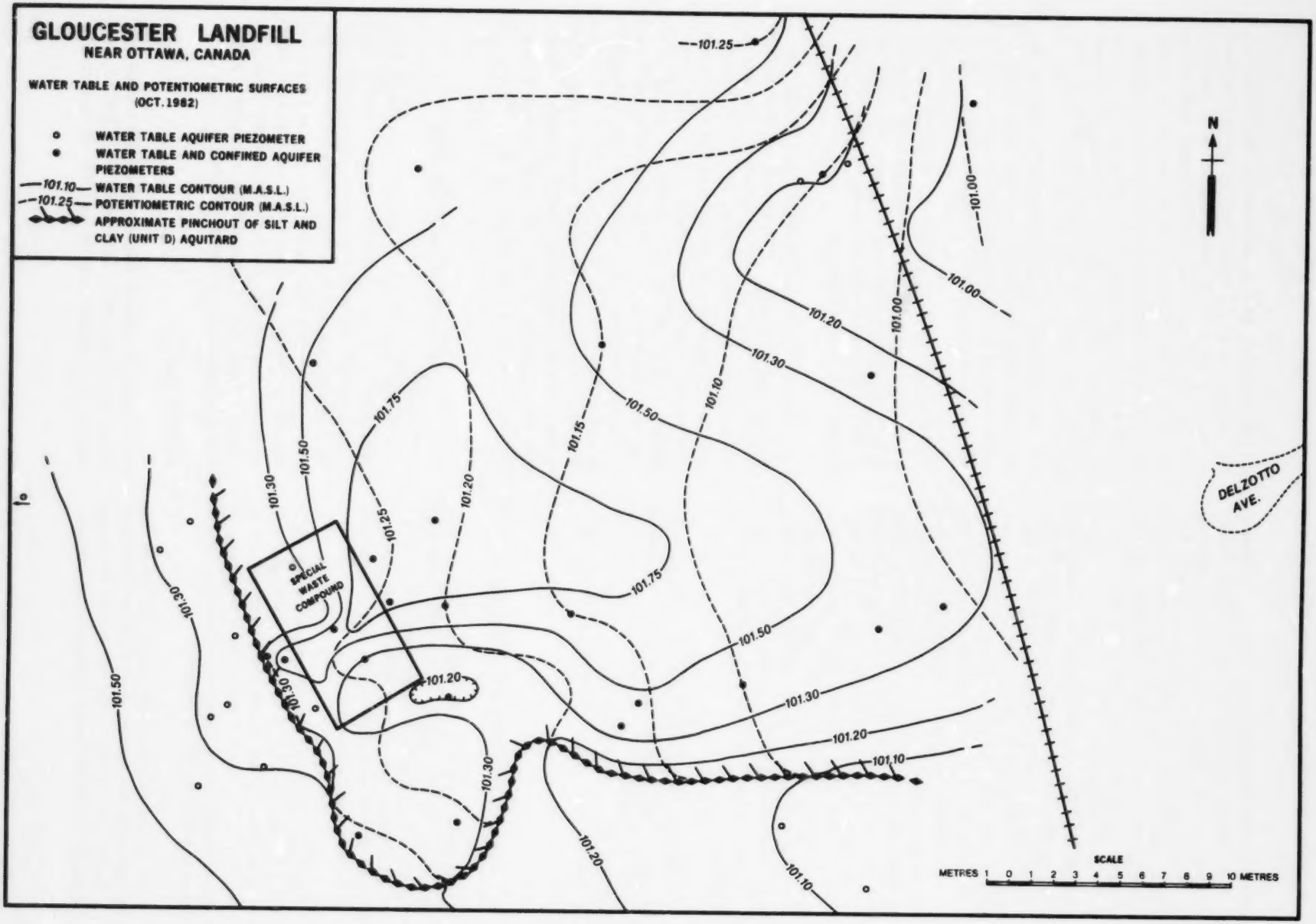

GLOUCESTER LANDFILL
NEAR OTTAWA, CANADA
WATER TABLE AND POTENTIOMETRIC SURFACES
(OCT. 1982)
WATER TABLE AQUIFER PIEZOMETER
WATER TABLE AND CONFINED AQUIFER PIEZOMETERS
101.10 WATER TABLE CONTOUR (M.A.S.L.)
101.25 POTENTIOMETRIC CONTOUR (M.A.S.L.)
APPROXIMATE PINCHOUT OF SILT AND CLAY (UNIT D) AQUITARD
N
SPECIAL WASTE COMPOUND
DELZOTTO AVE.
SCALE
METRES 1 0 1 2 3 4 5 6 7 8 9 10 METRES

Figure 23. Topography of the water table and the potentiometric surface of the outwash aquifer.

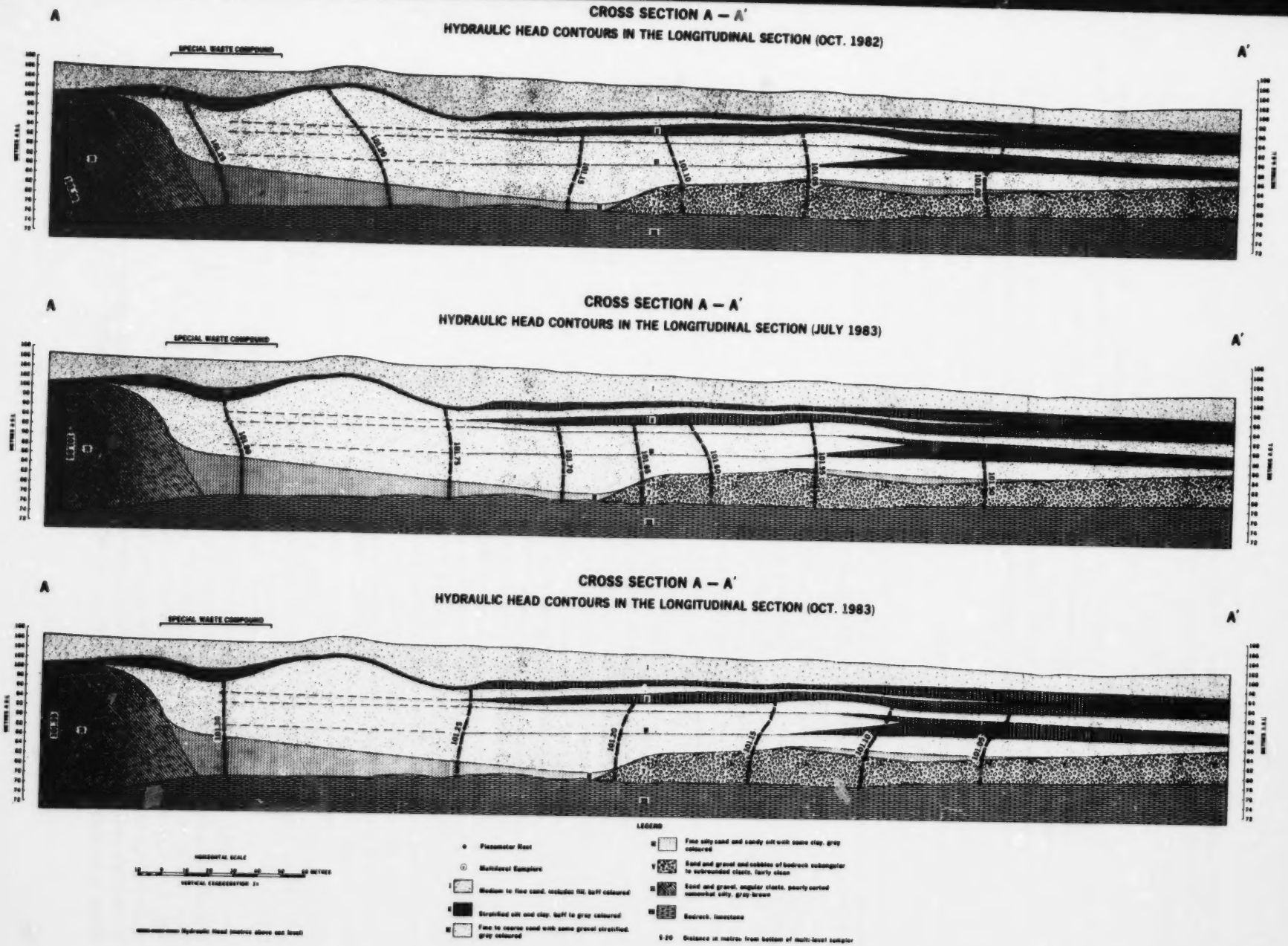


Figure 24. Hydraulic head contours along the A–A′ cross section during three periods in 1982 and 1983.

thoroughly mixed tracer in a closed volume of well screen is described by a differential equation, the solution of which is given by

$$V^* = - \frac{VOL}{At} \ln \frac{C}{C_0} \qquad (4.1)$$

where V^* = apparent velocity through the well screen (cm/day),
VOL = volume in which the dilution takes place,
A = cross-sectional area of measuring volume,
t = time,
C_0 = initial concentration of tracer (t = 0),
C = concentration of tracer at time t.

The apparent velocity, V^*, differs from the ground-water velocity in the undisturbed formation because the high permeability of the well screen distorts the flow pattern locally. To determine the ground-water velocity in the undisturbed aquifer, a correction factor, alpha (α), is applied to V^* and is dependent on the dimensions of the well screen and the hydraulic conductivity of the screen and the aquifer. From laboratory calibration tests and direct calculations, the value for alpha for well screens at the Gloucester Site is in the range 2.3-2.6.

The point dilution device used to conduct the velocity measurements is shown in Figure 25. This apparatus is similar to that described by Grisak *et al.* (1977) but uses an electrical conductance electrode to measure tracer concentration instead of a fluoride electrode. The device has three main components: an inflatable packer to isolate a well screen volume, an electrical mixing pump to mix the tracer, and a conductivity electrode in a flow cell to measure the tracer concentration. Tracer concentration is recorded at the surface by an electrical conductance meter. To conduct a point dilution test, the device is first lowered to the bottom of the piezometer. The injection line is then flushed to remove entrapped air, the packer is inflated and the mixing pump is started. When a stable background conductivity has been obtained, 15 mL of 70 g/L NaCl tracer solution is injected into the mixing circuit. After approximately 10 min, a peak concentration is reached, after which the time-varying electrical conductance is monitored for 6-10 h.

The time and electrical conductivity results are plotted semilogarithmically and, if all test conditions have been satisfied, produce a straight line. Equation 4.1 can then be solved for V^* either by using the plotted data directly or by linear regression to obtain values of C_0 and C. This apparent velocity, V^*, is then corrected to the actual ground-water velocity, V_{GW} (cm/day), by using either the equation

$$V_{GW} = \frac{V^*}{\alpha n} \qquad (4.2)$$

where n is the porosity (n = 0.35) or the use of a calibration curve (Fig. 26). An example of a dilution curve for a Gloucester piezometer is given in Figure 27.

In Figure 28, the results of the 20 point dilution tests in the lower aquifer are shown superimposed on the GTC hydrostratigraphic cross section. Ground-water velocities in the lower aquifer range from less than 1 to about 20 cm/day. This variation likely reflects the complex stratigraphy and sedimentology of the aquifer system and the presence of vertical flow components which modeling indicated could be significant. Considering all results for this flow system, an average velocity of 5.5 cm/day is indicated.

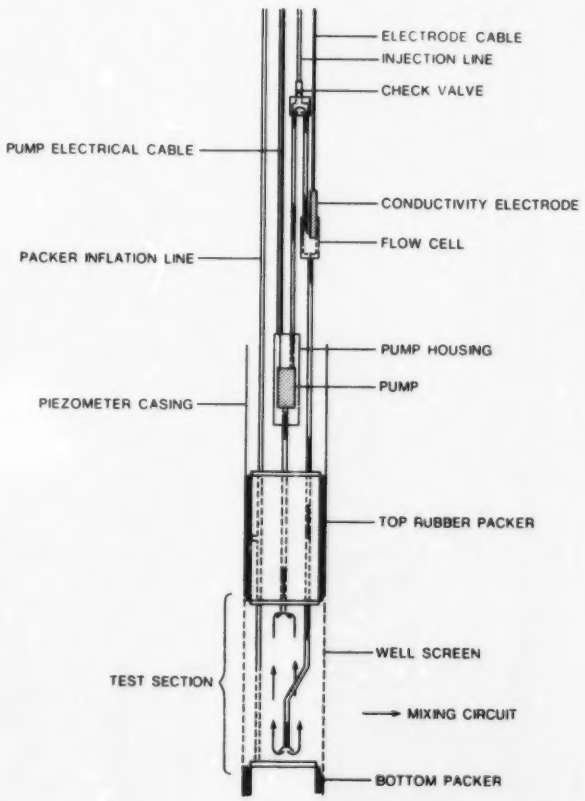


Figure 25. The point dilution device designed and built by D. Belanger.

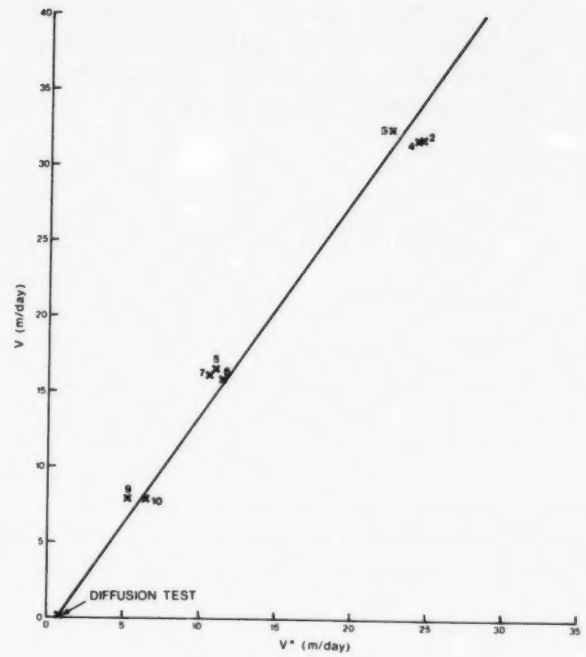


Figure 26. Velocity calibration curve.

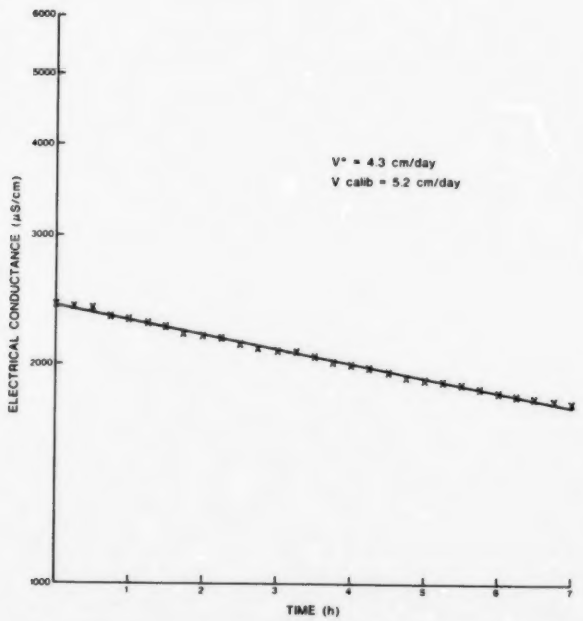


Figure 27. Dilution curve for 46P-3 observed during velocity test.

The upper aquifer at the Gloucester site was only tested in a few locations, since a low water table made many shallow piezometers unsuitable for velocity measurements. The measurements taken show the average velocity to be in the range from 2 to 3 cm/day.

In Table 10, velocities measured by point dilution are compared with velocities determined indirectly by Darcy's law calculations using pump-test and slug-test hydraulic conductivities and hydraulic gradients. Velocities calculated from pump-test hydraulic conductivities are, in most cases, higher than those determined by point dilution, whereas slug test results yielded velocities that are more similar to the point dilution values. Since the slug and point dilution tests are both carried out in a single piezometer, it is reasonable that they should give similar velocities. However, hydraulic conductivities calculated from pump test are more likely to represent an average value for the aquifer. If pump-test hydraulic conductivities are used, the average calculated velocity for the cross section including piezometers 51P-1, 55P-1 and 46P-1 is about 6 cm/day, which is in good agreement with the average lower aquifer velocity of 5.5 cm/day determined from point dilution data.

Table 10. Velocities Determined by Point Dilution, Pump Tests and Slug Tests* in (cm/day)

Piezometer	Point dilution	Pump test	Slug test
33-1	5.3	19.0	8.2
33-2	5.6	20.0	9.0
38-1	4.8	7.0	9.0
46-1	6.5	17.0	7.0
46-2	14.7	15.0	2.0
47-1	7.0	35.0	4.0
47-2	<1.0	18.0	0.3
55-1	5.2	7.0	6.0

*Hydraulic gradients are calculated for individual piezometers and range from 0.001 to 0.002. Porosity used was 0.35.

Bomb Tritium

About 125 tritium measurements have been made on ground-water samples from across the site by liquid scintillation counting at the Chalk River Nuclear Laboratories. The highest levels (>200 T.U.) occur in contaminated ground water beneath the Special Waste Compound and in the lower aquifer near 47P (Fig. 28). The high values in the contaminated water beneath the Compound are likely due to the presence of small quantities of tritium-labelled organic tracers that were present in some of the wastes. However, elevated activities in the lower aquifer near 47P are believed to represent waters that entered the

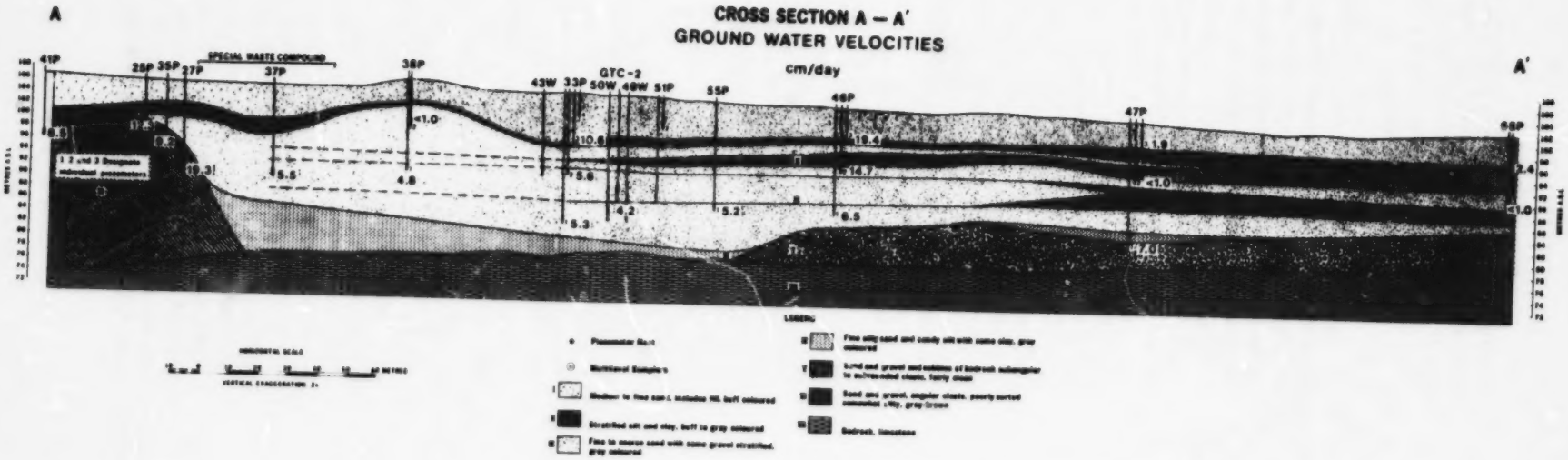


Figure 28. A–A' cross section showing ground-water velocities.

flow system in about 1963 when bomb tritium levels in atmospheric precipitation were at a maximum. Assuming that recharge occurred in the area immediately west of the Special Waste Compound (i.e. between wells 41W and 25W), an average ground-water velocity during the last 21 years of 5 cm/day is indicated. This value compares very favourably with velocities determined by other direct and indirect methods.

Ground-Water Geochemistry

MINERALOGY AND GEOCHEMISTRY OF THE AQUIFER MATERIALS

An X-ray diffraction analysis employing a Phillips diffractometer (Cu Kα radiation; $2\theta = 2°\text{-}66°$) was used to identify the major minerals in whole and sieved grain-size fractions of sediment samples representing the principal stratigraphic units at the Gloucester site.

Typical X-ray diffraction results are illustrated in Figures 29 and 30. Analysis of the data indicates that the mineralogy of all the unconsolidated units is similar and that quartz, feldspar (plagioclase, orthoclase and microcline) and carbonates (calcite and dolomite) are the dominant phases. The principal dark mineral is hornblende. Some samples show the presence of micas. However, there are apparently no significant occurrences of clay minerals, even in the finer units which contain clay-sized material. The lack of clay minerals is not surprising, considering the environment of deposition of the sedimentary sequence.

Whole and sieved grain-size fractions of sediments were examined under a binocular microscope. Approximate mineralogic proportions for the 60-80 mesh fraction of a sediment from a depth of 20 m at 49W (Fig. 17) are listed in Table 11. These results are typical for most sediments in the section and are comparable with the X-ray diffraction data.

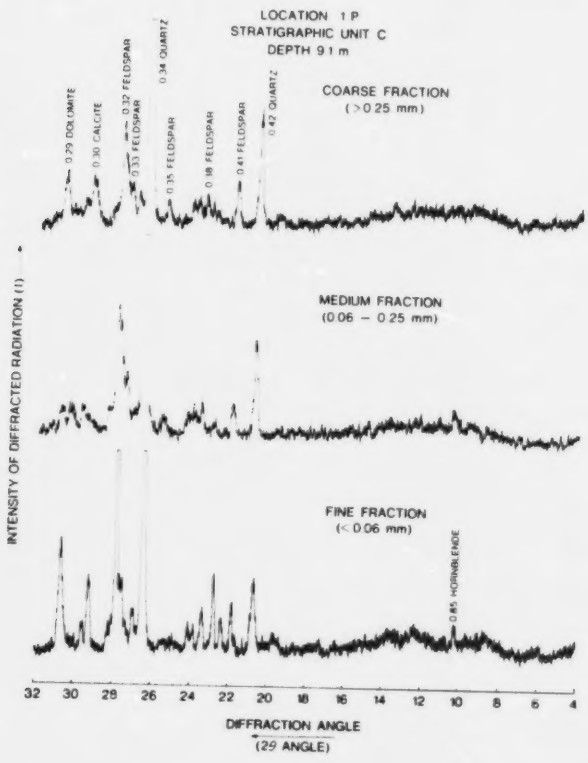


Figure 29. X-ray diffractogram for unit C.

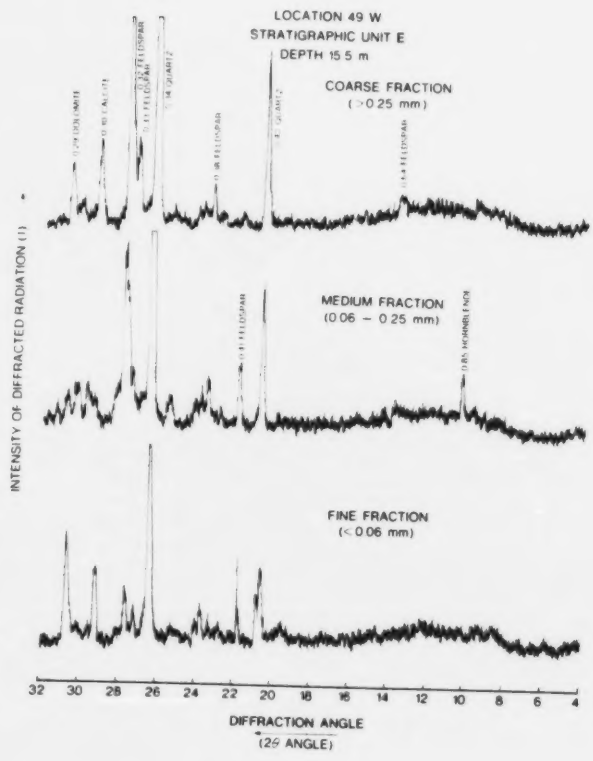


Figure 30. X-ray diffractogram for unit E.

Table 11. Optical Microscopic Analysis of 60–80 Mesh Fraction from Well 49W at 20-m Depth in Unit C

Mineral	Percent
Quartz	21
Feldspar	56
Mica	7
Calcite/dolomite	6
Hornblende	4
Garnet	3.5
Other	2.5
	100%

Optical examination of the sediments also revealed that trace amounts of detrital pyrite occur in the subaqueous outwash sequence (unit C). The presence of detrital pyrite, which indicates that the sediments are sedimentologically immature, is consistent with rapid deposition in a subaqueous environment adjacent to a glacial margin.

Table 12 lists results of analyses for sedimentary organic carbon determined using an Oceanography International total carbon analyzer with a Horiba PIR-2000 infrared detector. Organic carbon levels in unit E, the regressional sands and gravels, range from 0.01 to 0.35 percent by weight. The lowest values appear to occur in the recharge area near 41P. It is possible that the reservoir of sedimentary organic material has been reduced in this area by the percolation of well-oxygenated waters. Values in the silt and clay aquitard, unit D, are generally higher, ranging from 0.29 to 0.59 percent by weight O.C. Unit C,

Table 12. Results of Sedimentary Organic Carbon Analyses

Location	Depth (m)	Stratigraphic unit	Organic carbon (% wt.)	$\bar{x}$	σ
1	4.6	E	0.16	0.16	0.01
			0.15		
			0.16		
24	3.0	E	0.20	0.20	0.01
			0.21		
			0.20		
25	5.2	E	0.26	0.25	0.01
			0.24		
			0.24		
25	5.5	C–E	0.43	0.42	0.01
			0.41		
27	6.1	D	0.42	0.51	0.09
			0.51		
			0.59		
28	3.0	E	0.14	0.13	0.01
			0.12		
			0.14		

Table 12. Continued

Location	Depth (m)	Stratigraphic unit	Organic carbon (% wt.)	$\bar{x}$	σ
28	4.4	E	0.29	0.31	0.03
			0.30		
			0.35		
28	5.8	E	0.19	0.19	0.02
			0.20		
			0.17		
29	4.9	D	0.33	0.31	0.02
			0.30		
			0.29		
29	5.8	C	0.44	0.45	0.02
			0.47		
			0.44		
29	6.4	C	0.15	0.15	0
			0.15		
33	4.8	C	0.25	0.21	0.04
			0.21		
			0.18		
33	15.5	C	0.25	0.27	0.05
			0.33		
			0.23		
33	17.0	C	0.20	0.19	0.04
			0.23		
			0.15		
33	18.6	C	0.28	0.26	0.03
			0.24		
33	20.1	C	0.29	0.29	0.04
			0.33		
			0.26		
41	3	E	0.010	0.026	0.009
			0.017		
			0.030		
			0.034		
			0.032		
			0.035		
			0.027		
			0.025		
			0.024		
82	17.2	C	0.48	0.50	.05
			0.48		
			0.50		
			0.55		
			0.59		
			0.51		
			0.52		
			0.46		
			0.41		
82	17.5	C	1.04	0.85	0.24
			0.80		
			1.31		
			0.61		
			0.88		
			0.70		
			0.69		
			0.57		
			1.01		

the subaqueous outwash, shows considerable variation in organic carbon content, likely reflecting the sedimentologic complexity of the sequence. Maximum measured values, in some cases exceeding 1 percent by weight O.C., are associated with this unit.

INORGANIC CONSTITUENTS

Field Sampling Installations

Ground water for sampling and analysis was obtained from bundle-type multilevel samplers (Cherry *et al.*, 1983) or piezometers. Multilevel samplers were constructed by securing, with nylon ties, a bundle of up to 25 polyethylene tubes (6 mm O.D.; 1-mm wall thickness) along a PVC pipe (1.27 cm [0.5 in.] I.D.), which had a 30-cm screened section at the bottom (Fig. 31). The lower end of each polyethylene tube was wrapped with 100-mesh nylon screen (Nytex), and the screens were positioned at 1-m intervals along the support pipe. Installation was carried out by lowering the bundle and supporting tube into an open hollow stem auger, which was subsequently withdrawn, permitting the sediment to collapse around the assembly. Each polyethylene tube as well as the support pipe was then flushed using a portable peristaltic pump.

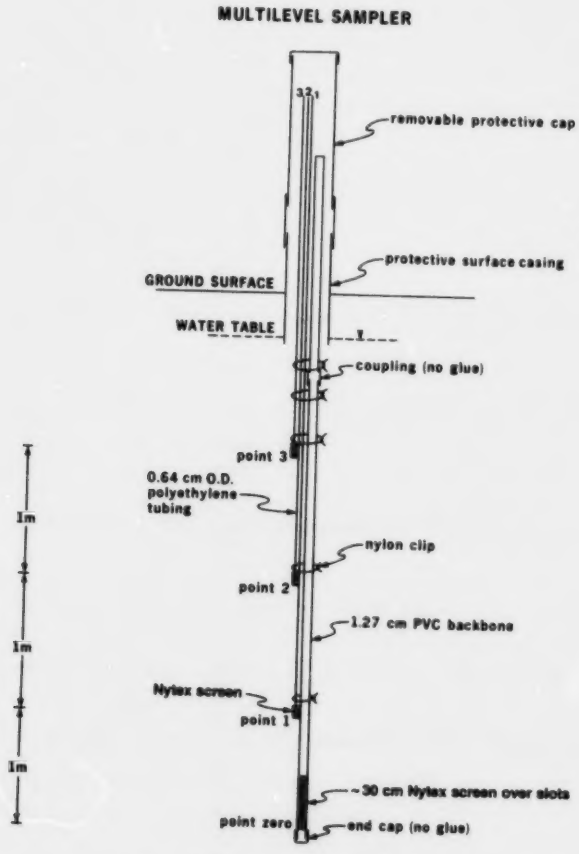


Figure 31. Design of multilevel sampler.

Piezometers were constructed from 3.2-cm (nests Nos. 1P–14P) or 5.04-cm (nests with Nos. greater than 14P) diameter PVC pipe. Screens were coiled or slotted (0.15 mm) (Johnston or Timco) and were 0.5 m long in the confined aquifer and 2 m in length in the water table aquifer. Installation was carried out by lowering an assembled unit into an open hollow stem auger, which was then withdrawn, permitting sediment to collapse around the piezometer. Following installation, the piezometers were developed by overpumping.

Sample Collection and Analysis

The general procedures for sample collection and analysis are outlined in Figure 32. Ground water for sampling was brought to the surface by a multichannel Masterflex peristaltic pump with silicone tubing in the pump head. For sampling multilevels, the individual polyethylene tubes in the bundle were attached directly to the pump or a specially designed sampling apparatus connected in series with the pump. Piezometers and the support pipes of multilevels were sampled by drawing water through a polyethylene tube, the lower end of which was positioned opposite the screened section of the well.

Measurements of ground-water temperature, specific conductivity (YSI models 32 and 33), pH, E_H and S^{2-}_{TOTAL} were conducted in the field in airtight flow cells. The pH and E_H were determined using Orion combination electrodes, and potentials were recorded on Orion 701 and 801A digital voltmeters. The pH electrode was calibrated frequently in two buffer solutions (ground-water temperature), which bracketed the range of values encountered in the ground water. Between individual readings, the electrode was standardized against a pH 7.00 buffer. The platinum combination electrode was stored in and calibrated against a ferrous-ferric solution in 1 *M* H_2SO_4 (Light, 1972).

Dissolved oxygen (DO) was determined using an Orbisphere Laboratories model 2606 oxygen indicator. Ground water was pumped directly through the probe and DO values were recorded when a stable level was achieved.

Sulphide measurements were performed using a glass/sulphide electrode couple in acidic (pH <5) solutions under closed, continuous flow conditions. The system was calibrated at 0.01 and 0.1 mg S^{2-}/L (Gulens, 1985).

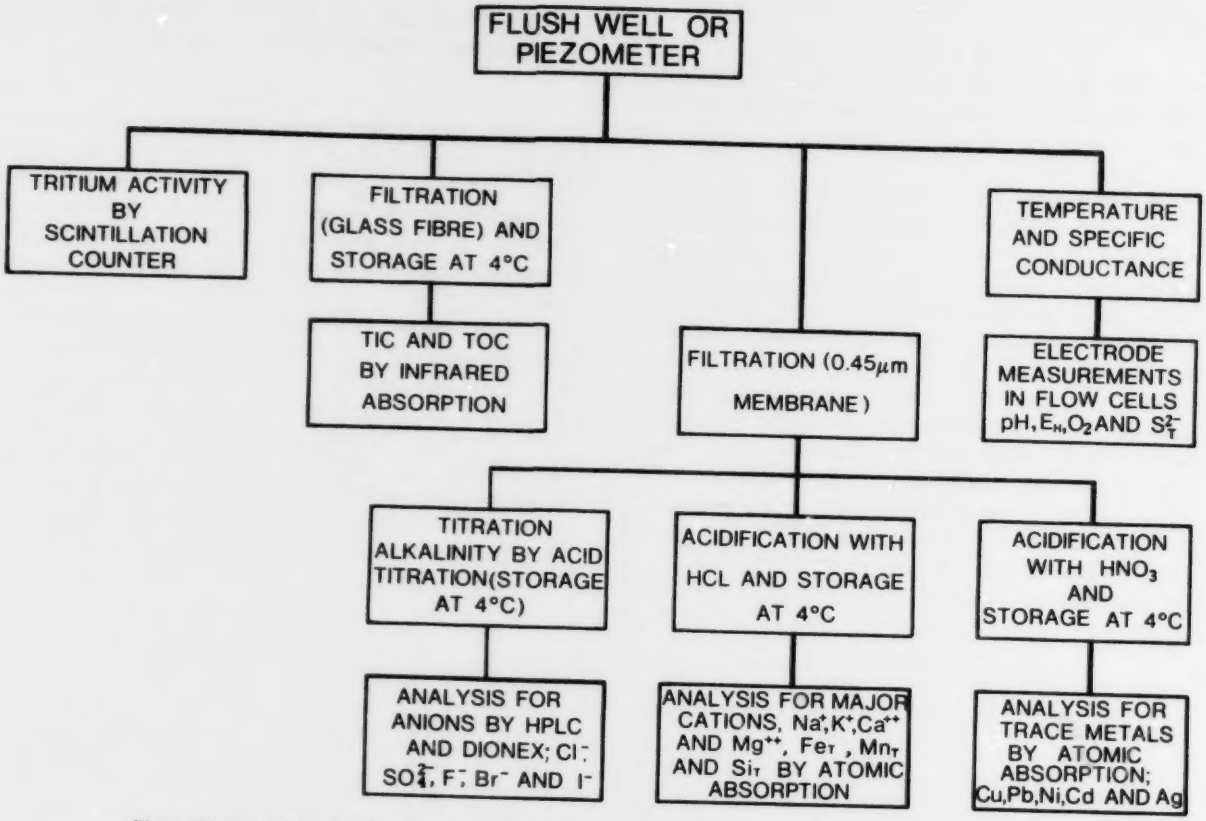

FLUSH WELL OR PIEZOMETER
TRITIUM ACTIVITY BY SCINTILLATION COUNTER
FILTRATION (GLASS FIBRE) AND STORAGE AT 4°C
TIC AND TOC BY INFRARED ABSORPTION
FILTRATION (0.45μm MEMBRANE)
TEMPERATURE AND SPECIFIC CONDUCTANCE
ELECTRODE MEASUREMENTS IN FLOW CELLS pH, E_H, O_2 AND S_T^{2-}
TITRATION ALKALINITY BY ACID TITRATION(STORAGE AT 4°C)
ACIDIFICATION WITH HCL AND STORAGE AT 4°C
ACIDIFICATION WITH HNO_3 AND STORAGE AT 4°C
ANALYSIS FOR ANIONS BY HPLC AND DIONEX; Cl^-, SO_4^{2-}, F^-, Br^- AND I^-
ANALYSIS FOR MAJOR CATIONS, Na^+, K^+, Ca^{++} AND Mg^{++}, Fe_T, Mn_T AND Si_T BY ATOMIC ABSORPTION
ANALYSIS FOR TRACE METALS BY ATOMIC ABSORPTION; Cu, Pb, Ni, Cd AND Ag

Figure 32. Flow chart showing procedures of sampling, preservation and analysis used at Gloucester Landfill.

The pump and all field instrumentation required for the electrode measurements were powered by a portable gasoline generator. Flow cells and instruments were protected from direct sunlight to minimize temperature fluctuations.

Depending upon the types of analyses desired, up to five ground-water samples were collected at each location when stable pH and E_H values were observed (usually after about 45 min of pumping). With the exception of the samples retained in the scintillation vials, all waters were passed through in-line filters prior to collection. The sample for dissolved inorganic and organic carbon analysis was filtered through a precombusted glass-fibre filter and collected in a precombusted (600°C) glass sample bottle. The other three were filtered through a 0.45-μm membrane (Millipore) and collected in acid-rinsed glass bottles. Two of the latter samples were preserved with acid (4 mL/L); one with HCl and the other with HNO_3. All samples were stored in the dark at 4°C following collection.

As soon as possible (within 12 h) after collection, 25 mL of water was withdrawn from the unacidified filtered (0.45 μm) sample for determination of alkalinity by acid titration. The rest of the sample was returned to dark storage at 4°C until analyses for anions could be performed by ion chromatography (Dionex model 12S): F^-, Cl^- and SO_4^{2-} with an electroconductivity detector and Br^- and I^- with an electrochemical detector.

The sample acidified with HCl was used to measure the concentrations of the major cations (Ca^{++}, Mg^{++}, Na^+ and K^+) and Fe_T, Mn_T and Si_T. Analyses were carried out by atomic absorption spectrophotometry (Perkin Elmer model 460).

The heavy metals—Cu, Pb, Ni, Cd and Ag—were measured by graphite furnace atomic absorption spectrophotometry (Perkin Elmer model 3058 and HGA-2000) using the sample preserved with HNO_3.

Analyses for TIC and TOC (total dissolved inorganic and organic carbon) were carried out using the sample collected in the precombusted bottle. A total carbon analyzer (Oceanography International) was employed for the measurements.

Results

The results of all inorganic analyses for 1980 and 1983 are listed in Appendix D. Table 13 contains 1982 data.

The distribution of chloride in plan and section in the confined aquifer is shown in Figures 33 and 34, respectively. Background chloride concentrations in ground waters from upgradient areas (i.e. west of the Special Waste Compound and municipal landfill) are less than 5 mg/L. Much higher levels occur beneath and downgradient from both the municipal landfill and Special Waste Compound. The vertical configuration of chloride contours along a section extending downgradient from the Special Waste Compound (Fig. 34) indicates that chloride originating in the Special Waste Compound is transported downward into the lower aquifer and then downgradient to the east and southeast. The leading edge of this chloride plume is currently past the end of Delzotto Road. Based on ground-water velocity measurements in the lower aquifer (Chapter 4) and assuming no retardation, the plume is likely advancing at a rate of about 5 cm/day.

The pH of ground waters upgradient from the waste site is on the order of 8.0. Since rainfall is generally acidic (pH <5), there is considerable neutralization during infiltration and percolation as a result of the dissolution of carbonate minerals which are a significant component in the unconsolidated sediments. Saturation index data (Table 14) indicate that all ground waters are near saturation with respect to calcite and dolomite. Within the chloride plume from the Special Waste Compound, the pH is generally less than 8.0 and frequently lower than 7.0 (e.g. 37M). The lowest pH value (5.78) was measured at 67M. The chloride plume also is characterized by lower O_2 and E_H values and increased sulphate and sulphide concentrations (up to 1.1 mg/L) and titration alkalinities. Measurable bromide occurs at several sampling locations, with the highest concentrations in the vicinity of the Special Waste Compound (e.g. locations 37M, 16M and 17M).

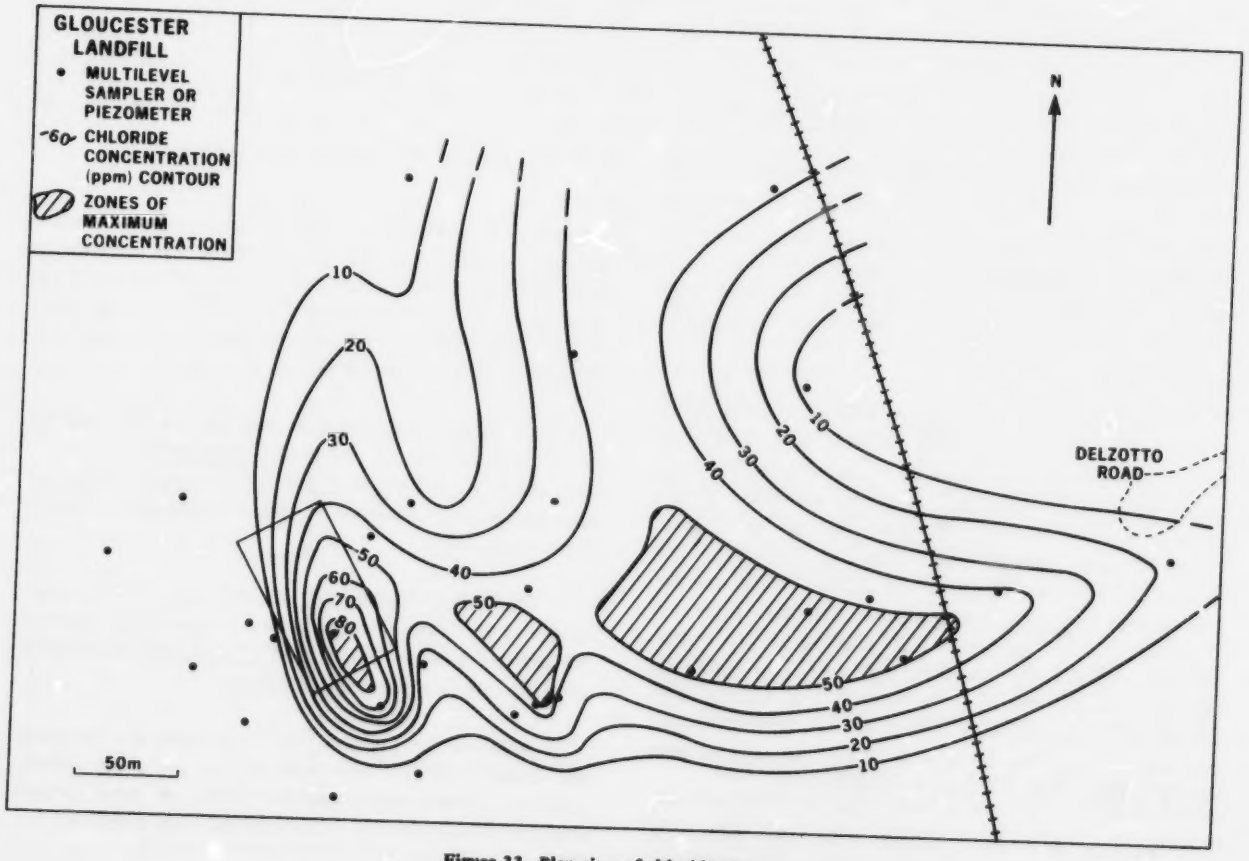


Figure 33. Plan view of chloride plume.

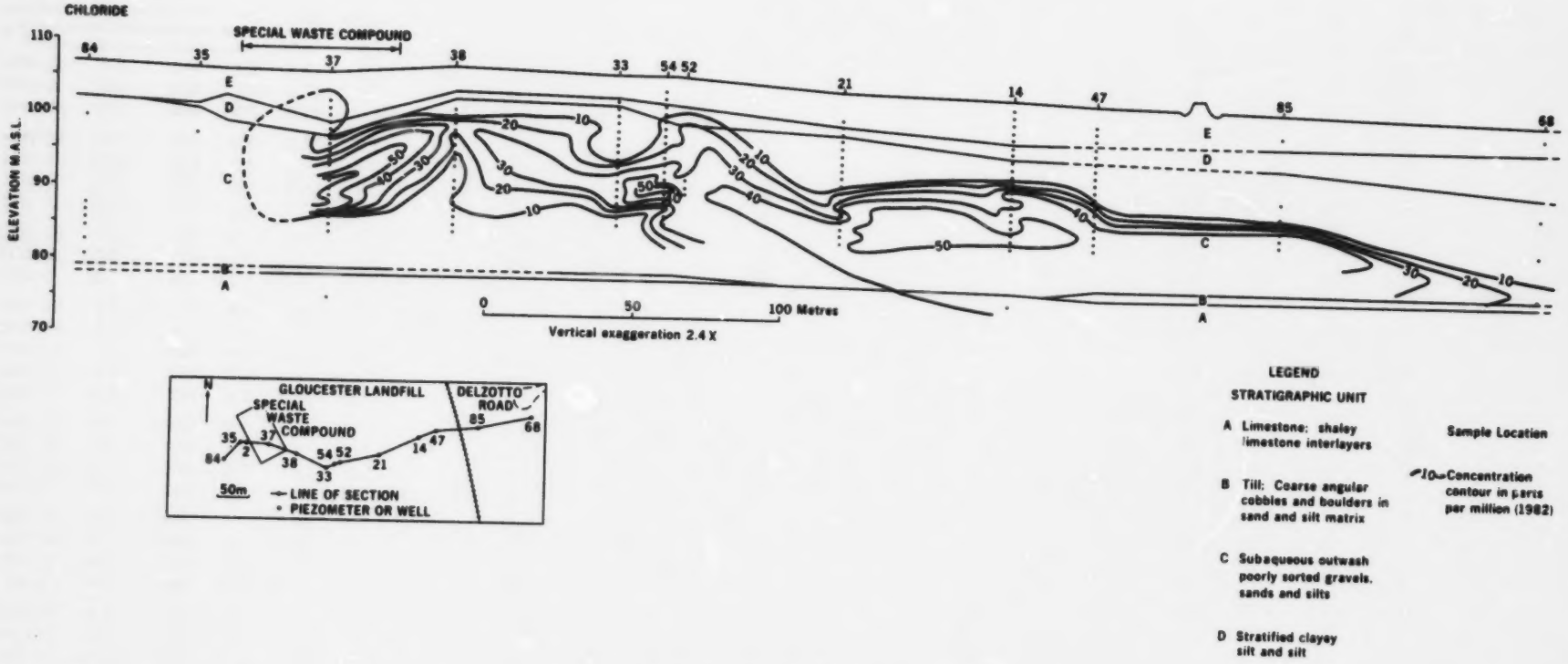


Figure 34. Cross-sectional view of chloride plume.

Table 13. Inorganic Chemical Analyses, 1982 Field Season

Site No.	Depth (m)	S.C.* (μS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (μg/L)	Tit. alk.† (meq/L)	TIC‡ (mg/L)	TOC§ (mg/L)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Fe (mg/L)	Mn (mg/L)	SO_4 (mg/L)	Cl (mg/L)	F (mg/L)	Br (mg/L)	Cu (μg/L)	Pb (μg/L)
1P-1	20.5	390	7.6	0.45	0.1	–	3.7	50	2.9	76	24	1.5	1.2	1.3	–	39	1.1	N.D.	N.D.	–	–
1P-2	10.5	270	8.0	0.19	0.1	–	2.6	36	1.0	44	13	1.2	0.8	0.9	–	23	0.3	N.D.	N.D.	–	–
1P-3	6.0	440	8.0	0.11	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
14M-1	21.3	830	7.8	0.24	–	10	3.7	49	2.0	108	40	2.8	2.3	0.5	0.06	143	72	0.8	N.D.	–	1.0
14M-5	17.3	740	7.7	0.04	0.3	170	5.3	65	21.4	122	35	9.9	2.6	1.8	0.04	2	135	2.4	N.D.	1.5	1.0
14M-18	4.3	108	8.7	0.42	0.2	–	0.9	12	0.9	19	3	1.0	0.6	1.5	–	6	1.0	0.3	N.D.	1.0	–
16M-4	18.2	307	7.9	0.15	0.1	–	3.6	44	1.9	56	18	1.5	1.2	1.0	–	35	3.4	0.1	N.D.	1.5	<1.0
16M-8	14.2	350	7.5	0.22	0.1	–	3.4	43	<0.6	57	14	1.2	1.1	0.5	–	27	1.3	0.1	N.D.	1.5	–
16M-12	10.2	430	7.7	0.17	0.2	–	4.7	63	3.0	66	17	4.0	1.3	1.5	–	10	2.3	0.1	N.D.	1.5	–
16M-16	6.2	730	7.3	0.13	0.1	–	7.0	99	14.2	120	20	13.0	6.7	1.5	–	65	27	0.8	1.2	1.0	2.3
18M-9	15.3	200	8.0	0.17	0.1	–	1.6	25	0.1	30	7	2.0	1.0	0.1	0.02	20	2.3	N.D.	N.D.	–	–
18M-14	10.2	220	8.0	0.03	0.1	–	1.7	27		33	10	3.2	1.1	0.1	0.02	26	4.1	N.D.	N.D.	–	–
18M-15	9.7	1000	7.4	0.16	0.1	–	4.0	59	1.8	101	29	40.5	2.0	0.6	0.04	82	46	1.2	N.D.	–	–
18M-19	5.3	1400	7.5	0.09	0.1	–	9.7	112	4.4	191	58	77.0	2.8	1.2	0.06	82	87	5.1	N.D.	–	–
19M-6	16.4	200	8.3	0.03	<0.1	–	1.7	–	–	26	7	2.6	1.4	0.1	0.02	9	1.8	N.D.	N.D.	–	–
19M-12	10.4	370	8.0	0.03	0.1	–	2.7	–	–	42	16	13.5	1.3	0.1	0.04	22	34	N.D.	N.D.	–	–
19M-16	6.4	850	7.4	0.01	<0.1	–	8.5	–	–	94	35	50.0	1.5	9.2	0.06	42	43	1.4	N.D.	–	–
19M-18	4.4	1020	6.4	0.09	0.1	–	12.4	–	–	153	28	9.9	0.9	98.0	0.61	40	18	1.4	N.D.	–	–
20M-1	18.0	250	8.2	0.01	<0.1	–	2.7	35	0.8	30	13	4.4	0.5	0.9	0.06	4	2.8	3.6	N.D.	–	–
20M-11	8.0	360	7.9	0.04	<0.1	–	3.8	50	7.1	47	17	8.0	1.5	4.0	0.15	3	10.5	2.2	N.D.	–	–
20M-12	7.0	1450	7.8	0.01	<0.1	–	16.0	151	160.2	220	73	35.5	3.5	17.5	0.34	N.D.	62	32.0	N.D.	–	–
20M-13	6.0	–	–	–	–	5	–	–	14 430	61	20	1.9	0.8	<0.1	<.01	33	8.7	1.4	N.D.	–	–
20M-15	4.0	810	6.9	0.09	0.1	N.D.	10.0	6.5	15.3	108	15	15.0	15.3	67.0	0.38	N.D.	12.0	0.5	N.D.	–	–
21M-1	20.0	1860	7.4	0.14	0.1	10	7.9	98	3.9	260	9	140	4.5	1.7	0.09	414	168	16.2	N.D.	–	–
21M-9	12.0	720	7.6	0.14	0.4	5	6.5	83	1.1	118	39	16.0	1.7	0.1	0.06	70	28	2.7	N.D.	–	–
21M-16	5.0	200	8.2	0.35	0.1	–	1.8	21	<0.6	30	8	1.7	0.8	<0.1	<0.01	15	2.8	1.8	N.D.	–	–
33M-1	21.0	370	7.8	0.15	0.1	–	3.6	54	<0.7	46	19	2.4	1.4	<0.1	0.04	33	4.1	1.0	N.D.	–	–
33M-6	16.0	1080	7.4	0.15	0.2	–	10.9	144	7.4	153	49	40.5	3.5	1.0	0.07	34	107	2.1	1.9	–	–
33M-12	10.0	210	8.4	0.32	0.2	–	1.6	26	<0.7	30	9	2.5	0.9	<0.1	0.04	14	6.8	0.5	N.D.	–	–
33M-18	4.0	290	7.8	0.32	5.9	–	3.1	47	<0.7	43	12	2.6	1.1	<0.1	0.04	16	2.7	1.4	1.9	–	–
34M-3	18.8	986	7.6	0.38	0.1	–	8.0	112	33.6	162	48	4.7	3.0	2.4	0.06	36	140	16.0	N.D.	–	–
34M-7	14.8	643	7.8	0.18	0.1	–	5.1	70	3.4	85	24	3.1	2.1	0.4	0.06	7	68	0.5	N.D.	–	–
34M-17	4.8	744	7.2	0.20	0.1	–	7.9	98	2.1	116	33	23.0	6.3	12.5	0.40	179	13	1.9	N.D.	–	–

* Specific electrical conductance at T ≈ 10°C.
† Titration alkalinity.
‡ Total inorganic carbon.
§ Total organic carbon.
Note: N.D. means not detected; dash means not measured.

Table 13. Continued

Site No.	Depth (m)	S.C. (μS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (μg/L)	Tit. alk. (meq/L)	TIC (mg/L)	TOC (mg/L)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Fe (mg/L)	Mn (mg/L)	SO_4 (mg/L)	Cl (mg/L)	F (mg/L)	Br (mg/L)	Cu (μg/L)	Pb (μg/L)
37M-1	22.0	420	8.0	0.09	0.1	–	3.6	47	<0.6	46	18	2.2	1.6	0.3	0.02	49	2.8	1.2	N.D.	–	–
37M-6	17.0	890	7.3	0.09	<0.1	–	6.1	87	7.5	140	35	19.0	2.3	2.7	0.14	116	190	1.3	58	–	–
37M-9	14.0	1380	7.2	0.13	0.2	–	10.4	141	61.0	210	58	87.5	3.5	6.7	0.15	287	200	4.2	48	–	–
37M-18	5.0	1430	7.2	0.19	0.3	–	7.6	102	11.4	290	66	24.0	2.2	0.7	4.00	346	70	14.9	16	–	–
41P-1	10.0	380	7.9	0.55	2.1	–	3.6	47	1.3	60	21	1.2	0.9	0.5	0.14	32	6.2	N.D.	N.D.	–	<1
47M-2	18.6	570	7.9	0.14	0.15	–	2.9	38	1.5	83	27	2.0	2.0	1.0	–	59	70	0.4	N.D.	1.0	–
47M-6	14.6	603	7.8	0.17	0.1	–	4.1	54	1.2	100	24	2.3	1.6	1.0	–	90	45	0.6	N.D.	1.0	1.0
47M-17	3.7	197	8.0	0.44	0.5	–	1.8	23	1.8	32	6	1.5	1.0	0.5	–	14	1.3	0.4	N.D.	1.5	<1
49W	17.0	1085	–	–	–	–	–	–	7.5	123	37	8.9	2.7	0.5	–	98	80	1.34	1.4	1.5	1.0

In 1980 and 1982, ground-water samples were analyzed for iron, manganese and trace metals. Trace metals were low at all locations near the Special Waste Compound and relatively high iron and/or manganese concentrations (above 5 ppm and/or 1.0 ppm, respectively) were only apparent at locations 16M, 18M, 19M, 20M, 34M and 37M (Fig. 10). Since the potentially toxic metals were not apparently being mobilized and transported, subsequent ground-water samples were not analyzed for these components.

Table 14. Saturation Indices for Mineral Phases from WATEQF Analysis of Data from Table 13

	Mineral phase							
Site No.	$CaCO_3$ CALCITE	$CaMg(CO_3)_2$ DOLOMITE	$Fe(OH)_3$ FEOH3A	$CaSO_4$ GYPSUM	$FeCO_3$ SIDERITE	FeS FESPPT	FeS MACKIT	FeS_2 PYRITE
1P-1	0.2	-0.1	3.6	-1.7	-4.0	–	–	–
1P-2	0.2	0.1	3.2	-2.1	-0.5	–	–	–
14M-1	0.5	0.8	-4.0	-1.1	0.0	12.8	13.5	17.2
14M-5	0.6	0.8	-4.0	-2.9	0.6	14.4	15.2	19.5
14M-18	0.2	-0.2	3.3	-2.9	-6.6	–	–	–
16M-4	0.4	0.4	3.1	-1.8	0.3	–	–	–
16M-8	0.0	-0.4	2.9	-1.9	-0.4	–	–	–
16M-12	0.4	0.4	3.1	-2.3	0.4	–	–	–
16M-16	0.4	0.2	1.3	-1.4	-0.4	–	–	–
18M-9	-0.1	-0.6	0.7	-2.2	-1.0	–	–	–
18M-14	0.0	-0.4	0.1	-2.1	-1.0	–	–	–
18M-15	0.1	-0.2	-0.2	-1.3	-0.3	–	–	–
18M-19	0.7	1.0	0.9	-1.2	0.3	–	–	–
19M-6	0.2	-0.2	0.5	-2.6	-0.8	–	–	–
19M-12	0.2	0.1	-1.0	-2.1	-0.9	–	–	–
19M-16	0.3	0.3	0.1	-1.7	1.1	–	–	–
19M-18	-0.3	-1.3	-0.4	-1.6	1.3	–	–	–
20M-1	0.3	0.5	1.6	-3.0	0.6	–	–	–
20M-11	0.3	0.4	1.9	-2.9	1.1	–	–	–
20M-12	1.3	2.2	1.7	–	2.0	–	–	–
20M-15	-0.0	-0.8	0.9	–	1.6	–	–	–
21M-1	0.7	1.0	-4.1	-0.5	0.3	12.7	13.5	17.8
21M-9	0.5	0.7	-4.3	-1.3	0.0	12.2	12.9	16.7
21M-16	0.1	-0.2	2.0	-2.3	-5.2	–	–	–
33M-1	0.2	0.2	1.5	-1.9	-1.0	–	–	–
33M-6	0.6	0.9	1.6	-1.6	0.3	–	–	–
33M-12	0.4	0.3	1.0	-2.4	-5.1	–	–	–
33M-18	0.2	-0.1	2.1	-2.2	-3.5	–	–	–
34M-3	0.8	1.2	3.8	-1.6	-2.0	–	–	–
34M-7	0.6	0.8	2.8	-2.4	-0.3	–	–	–
34M-17	0.1	-0.3	2.9	-1.0	0.9	–	–	–
37M-1	0.4	0.5	1.8	-1.8	-0.1	–	–	–
37M-6	0.3	0.1	0.7	-1.1	0.4	–	–	–
37M-9	0.5	0.5	1.5	-0.7	0.8	–	–	–
37M-18	0.5	0.4	1.5	-0.5	-0.3	–	–	–
41P-1	0.3	0.3	3.0	-1.8	-7.0	–	–	–
47M-2	0.4	0.5	2.9	-1.5	0.2	–	–	–
47M-6	0.5	0.5	3.0	-1.3	0.1	–	–	–
47M-17	0.0	-0.5	3.2	-2.4	-5.1	–	–	–
49W	0.6	0.9	1.9	-1.2	0.1	–	–	–

Note: Saturation index = log (ion activity product) minus log (solubility product).

ORGANIC CONSTITUENTS

Sample Collection

Most ground waters for organic analysis were obtained from the multilevel samplers. Field tests at 56M indicated that the polyethylene tubing used in the construction of the samplers was suitable for collecting waters containing volatile organics. This multilevel was assembled with duplicate sets of polyethylene and Teflon tubing. Comparison of analytical data (Table 15) for corresponding water samples from the two types of tubing indicated similar performance for both materials.

Multilevel samplers installed in 1979 (16M to 21M and 23M) as well as three completed in the winter of 1982 (67M, 68M and 69M) had a PVC backbone in which the joints were glued with PVC or CPVC cement. Laboratory tests indicated that these cements contain volatile organic compounds (Table 16). Some of these organic components, especially THF, have been found in ground-water samples from multilevel points adjacent to glued joints. As a result, all subsequent installations were constructed without glue.

Table 15. Comparison between Tubing Types at 56M-4 (average of four replicates) (μg/L)

Compound	Teflon	Polyethylene	Percent difference*
Dichloromethane	8.75	6.6	-25
1,1-Dichloroethylene	4.06	3.83	-6
1,1-Dichloroethane	89.6	93.5	+4
Trans-1,2-dichloroethylene	34.4	32.9	-4
Chloroform	1200.	1320.	+10
1,2-dichloroethane	146.	158.	+8
1,1,1-Trichloroethane	72.2	72.9	+1
Carbon tetrachloride	94.7	96.7	+2
Trichloroethylene	54.5	53.8	-1
Benzene	210.	227.	+8
Bromoform	5.7	5.9	+4
1,1,2,2-Tetrachloroethane	1.75	0.9	-49
Toluene	232.	268.	+16
Chlorobenzene	2.75	2.5	-9
Ethylbenzene	3.16	4.08	+29

*Teflon values = 100%.

Table 16. Compounds Detected in PVC Cements by GC/MS

CPVC cement	PVC cement
Dichloromethane	Dichloromethane
2-Propanone	2-Propanone
THF (MM)	THF (MM)
Butanone (M)	Butanone (M)
Chloroform	Benzene
Tetrahydromethylfuran	Dipropyl ether
Benzene	Cyclohexanone (M)
Tetrahydrodimethylfuran	2-Hexanone
Propylacetic acid	Toluene
Cyclohexanone (M)	
Toluene	

MM – Very large quantities.
M – Major.

Most of the ground-water samples for volatile organic analysis were collected using a specially built volatile organic sampler (VOS) based on an earlier design by J. Barbash, University of Waterloo (Fig. 35). The VOS, fabricated entirely from stainless steel and Teflon, was constructed to minimize the potential for loss of volatiles during sampling due to volatilization and sorption.

To collect a sample in the field, the inlet tube of the VOS was connected to a point on a multilevel piezometer and the outlet to a peristaltic pump. Water from the multilevel point was pumped to waste through the bypass route of the VOS for about 10 min at a rate of approximately 300 mL/min. A glass vial was then screwed into the Teflon block and the three-way valve positioned to produce a vacuum in the vial. The toggle valve was then opened to permit water to pass through the glass sample vial. After about 1 min, the flow was again directed through the bypass route and the glass vial was removed, emptied and rethreaded into the Teflon block. This rinsing procedure was repeated four times to ensure that the system and vial were adequately flushed. After the fourth filling, the vial was removed and immediately sealed with a Teflon-lined septum held in place by a screw cap. Care was taken to exclude bubbles from the vial when sliding the septum through the meniscus of the sample. Glass vials containing samples were stored in the dark at 4°C prior to analysis.

In the fall of 1983 at locations 83M, 84M, GTC1 and GTC2, an improved apparatus (Fig. 36) was introduced to collect samples for volatile organic analysis. With this system, water was drawn from a multilevel tube by a peristaltic pump and passed through a three-way Teflon valve which accepts a 50-mL polypropylene syringe fitted with a Teflon on-off valve. To collect a sample, ground water was pumped to waste through the three-way valve for a period of about 10 min. The pump speed was then reduced and the three-way valve positioned to direct water into the syringe. When the desired volume had entered the syringe, the three-way valve was rotated to permit the syringe to be emptied into the stream passing to waste. This procedure was repeated a second time to ensure that the valve and syringe were adequately flushed. The syringe was then filled a third time and the on-off valve at the tip closed. The filled syringe and on-off valve were removed from the sampler and stored in the dark at 4°C for a period not exceeding 30 h. Laboratory tests indicated that after this length of time changes in the

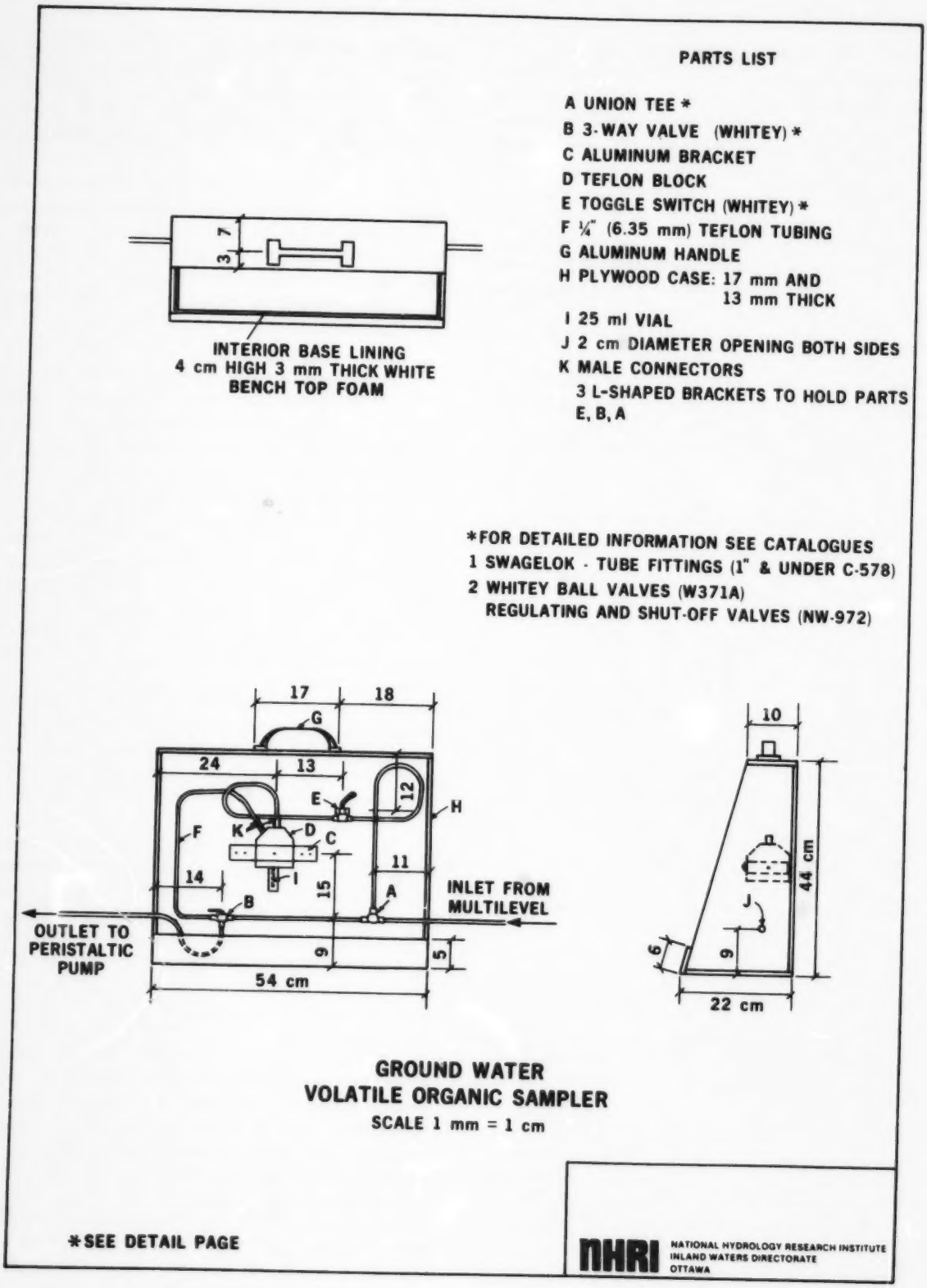


Figure 35. Volatile organic sampler designed and built by M. Priddle and B.W. Graham.

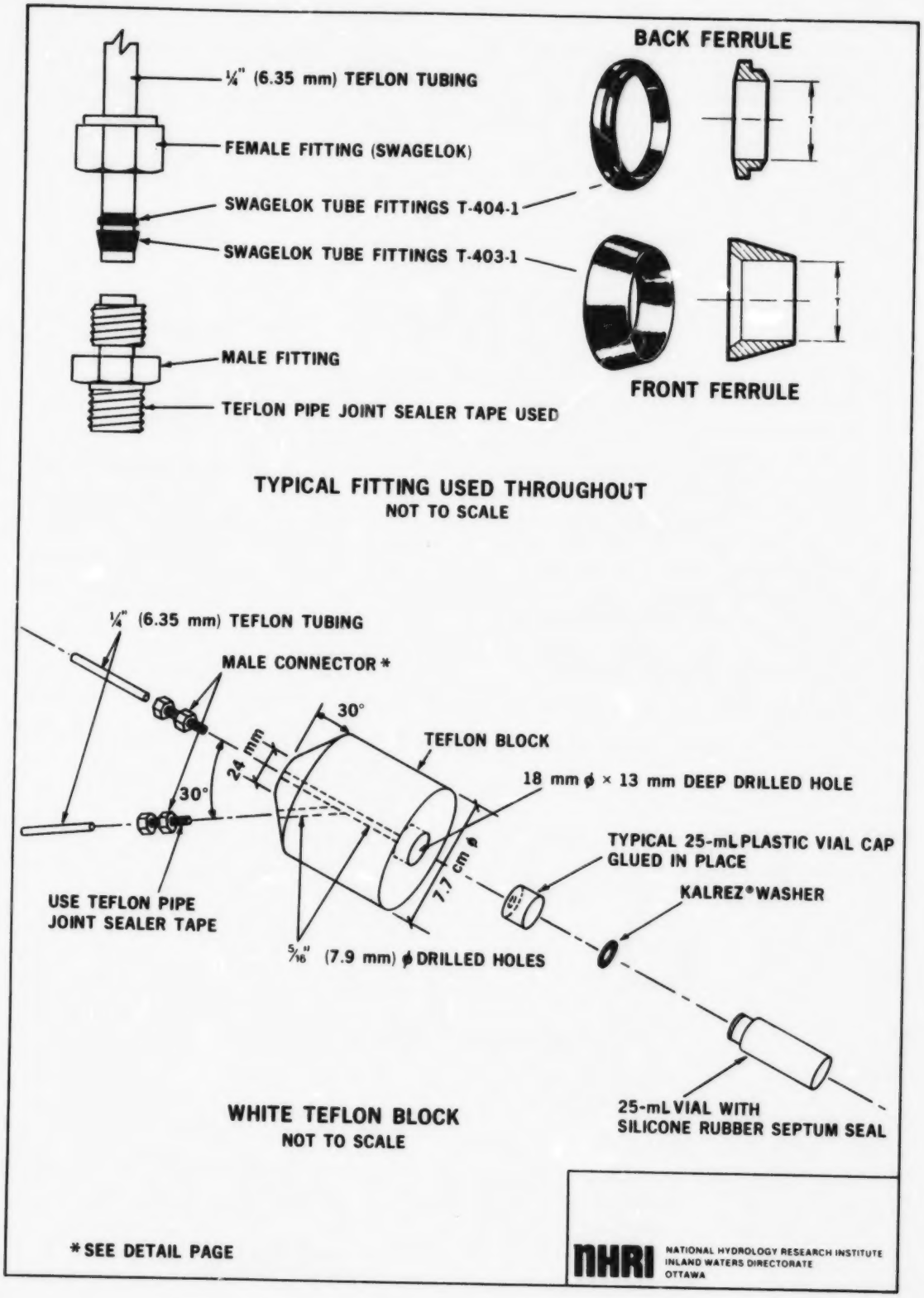
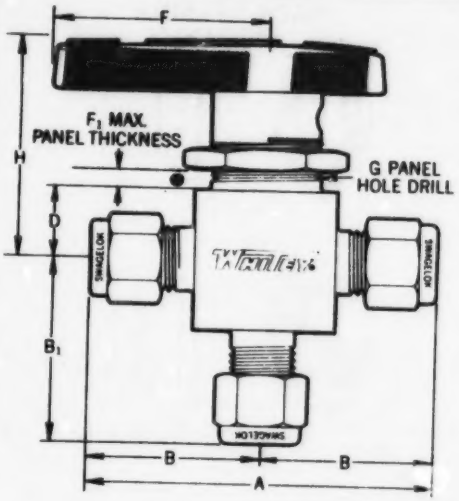
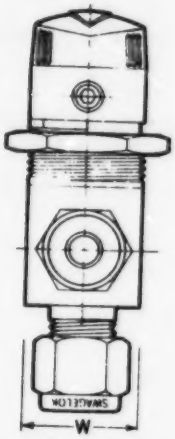
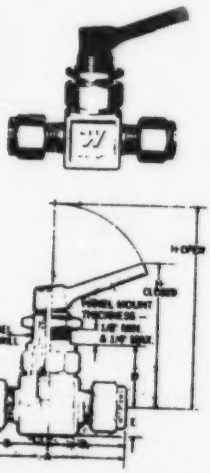


Figure 35. Continued.

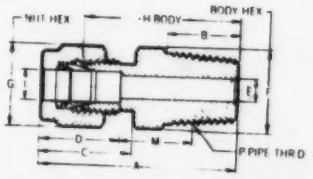


3-WAY BALL VALVE

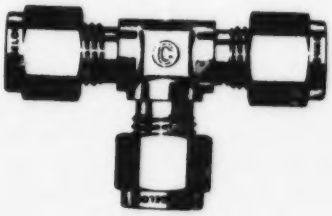


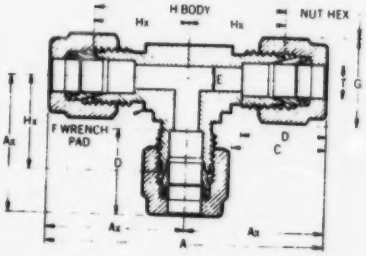


TOGGLE-OPERATED
FORGED BODY
SHUT OFF VALVES



MALE CONNECTOR-Fractional Tube to Male NPT Thread





UNION TEE-All Fractional Tube

PARTS LIST

QTY	CATALOGUE No.	BRAND NAME	PART
1	400-3	SWAGELOK	UNION TEE
2	400-1-4	SWAGELOK	MALE CONNECTOR
1	43XS4	WHITEY	3-WAY BALL VALVE
1	1GS4	WHITEY	TOGGLE VALVES

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Figure 35. Continued.

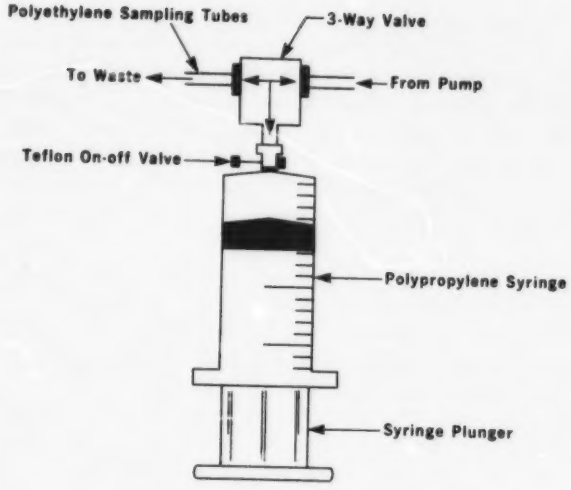

Figure 36. Farrington, Lockwood syringe sampler.

concentrations of the volatile compounds present near locations 33M, 52M, 53M and 54M could be significant due to leaching from and uptake by the sample container (Farrington, Lockwood Co. Ltd., 1983).

Analytical Methods

Routine analysis of volatile organics in ground-water samples was done by purge and trap gas chromatography (Hewlett Packard 5830 A). Confirmation of contaminants in certain selected samples was accomplished by gas chromatography/mass spectrometry employing a Finnigan 4000 GC/MS system with an attached and dedicated data system. These methods are outlined in Figure 37.

In the purge and trap GC analysis procedure, a 5-mL aliquot of the sample is purged with high purity N_2 gas at 40 mL/min for 12 min. Transfer of sample from the glass vials (collected using the VOS) to the GC sparge tube was accomplished by pipetting. When the improved sampling system incorporating syringes was introduced, the transfer was carried out by securing a Luer Lock needle to the Teflon on-off valve on the syringe tip and injecting the sample through a septum directly into the sparge tube. This procedure eliminated a step in which exposure of the sample to air was unavoidable.

The volatile organics removed from the sample are trapped on a silica gel/Tenax column, from which

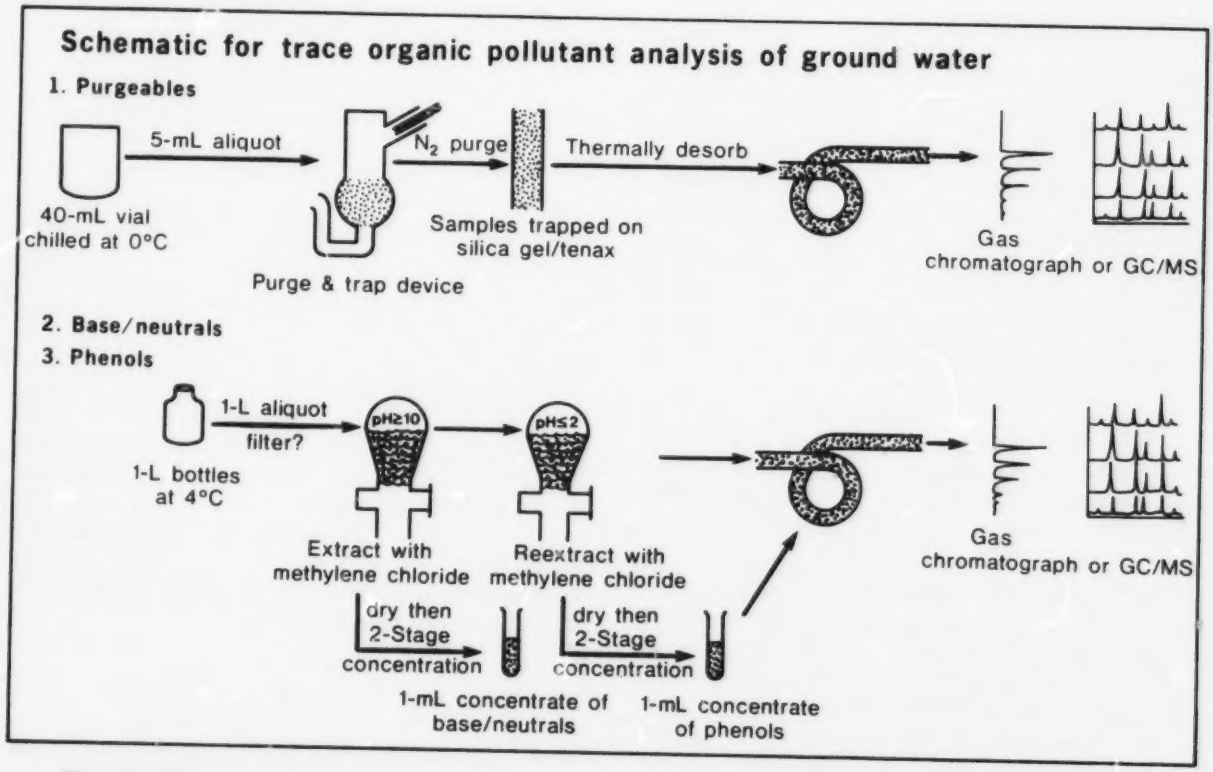

Figure 37. Schematic representation of gas chromatographic/mass spectrometric (GC/MS) analysis of pollutants in ground water.

they are thermally desorbed at 250°C for 10 min with 30 cm^3 N_2/min. The volatile organics are chromatographically separated on a Supelco 1% SP-1000 60/80 mesh, Carbopak B, stainless steel column 8 ft × 1/8 in., and are then measured with a flame ionization detector (FID). Identification and quantification were done by comparing sample spectra with those for diluted, commercially prepared, external standards with concentrations of 40 μg/L. The FID response was found to be linear over the range of concentrations encountered in the samples.

The GC/MS analyses were conducted on the volatile or purgeable fraction of the Gloucester ground waters as well as on the base/neutral and acid or phenol fractions which were separated by liquid-liquid extractions (Fig. 37). The extracts were chromatographically separated on a 30-m fused silica bonded-phase capillary column. Instrument response was checked by using an internal standard (d-10 anthracene), and retention times were monitored using external standards.

All glassware used in the analytical procedure was washed with soap and water, rinsed successively with tap water, distilled water, acetone, and hexane and then combusted at 350°C for at least 7 h.

Volatile Organic Contaminants

The results of analysis for volatile organic compounds by GC and GC/MS are listed in Appendix E. Pairs of compounds which coelute (e.g., 1,1-dichloroethane and tetrahydrofuran) are underlined in the column headings. For coeluting pairs, concentration values for each sampling point have been listed under the compound which is felt most likely to be present or to be predominant. Decisions concerning which compound is likely or predominant were based on extrapolations from the few locations where GC/MS confirmation data (Appendix E and Figs. 38 and 39) are available and on patterns of association with identifiable components.

The areal distributions of several organic compounds in the vicinity of the Special Waste Compound are illustrated in Figure 40. For each location, the value contoured is the maximum concentration encountered in section.

As with chloride, contamination to the north and east of the compound could be a result of pollution from the municipal landfill operation rather than transport from the Special Waste Compound. Consequently, the positions of the northern boundaries of the plumes shown in Figure 33 may exaggerate the extent of contaminant migration from the compound. Nevertheless, the distribution patterns indicate that contaminant migration from the compound is occurring to the east and southeast, and that the extent of transport varies for different compounds. Figure 41 shows the configurations of the plumes in greater detail along a section oriented approximately east-west. It is apparent that the contaminants are entering the confined, lower aquifer near the Special Waste Compound and are being transported downgradient (Fig. 41) at depth in this flow system.

The differences in the extent of migration for the various compounds are also clearly illustrated in the sections. For example, 1,4-dioxane has migrated more than 250 m, whereas carbon tetrachloride is still limited to the Special Waste Compound area. Some of the differences in the extent of transport undoubtedly reflect the input history and distribution for the various compounds. However, the major controlling process appears to be chromatographic dispersion. The distance and, hence, rate of migration correlate inversely with the compound's octanol-water partition coefficient (log K_{ow} is given for each compound in Figs. 40 and 41), which is a measure of the degree of retardation expected in a ground-water flow system (Equations 2.2 and 2.6). Chlorinated aliphatics (carbon tetrachloride and 1,2-dichloroethane) and aromatics (benzene), which have relatively high coefficients, are apparently much less mobile in the aquifer system than the water soluble, oxygen-containing substances (Fig. 41) such as 1,4-dioxane, with log K_{ow} values less than 1.

By rearrangement of Equations 2.2 and 2.6, it can be shown that for a given sedimentary organic carbon content

$$R_f - 1 = B\ (K_{ow})^{0.72} \qquad (5.1)$$

where B is a constant. Thus, for a particular organic compound, z,

$$(R_f^z - 1) = B\ (K_{ow}^z)^{0.72}$$

or

$$\log (R_f^z - 1) = 0.72 \log K_{ow}^z + \log B \qquad (5.2)$$

Comparison of Figures 41 and 34 indicates that the plumes for the more mobile organics are very similar in form to that for chloride, although less extensive. If it is assumed that all compounds were disposed of simultaneously and that the retardation factor for chloride is 1 (i.e. no retardation), the retardation factor for an organic compound, z, could be estimated from:

$$R_f^z = \frac{T_{Cl}}{T_z} \qquad (5.3)$$

where T_{Cl} is the length of the chloride plume and T_z is the length of the particular organic compound plume.

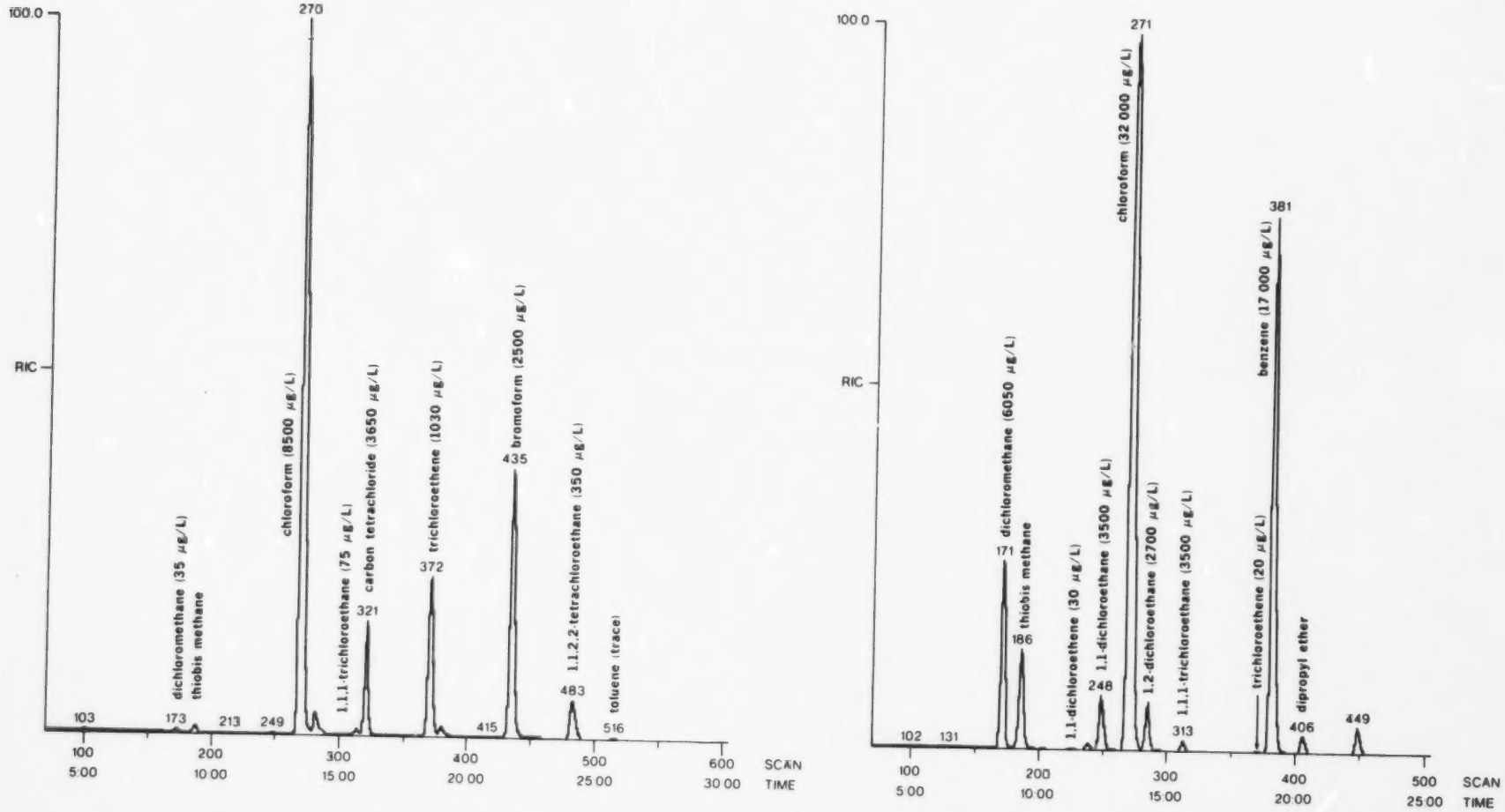

Figure 38. GC/MS analysis (reconstructed ion current chromatograms) of volatile organic compounds in two samples of ground water from multilevel 37M.

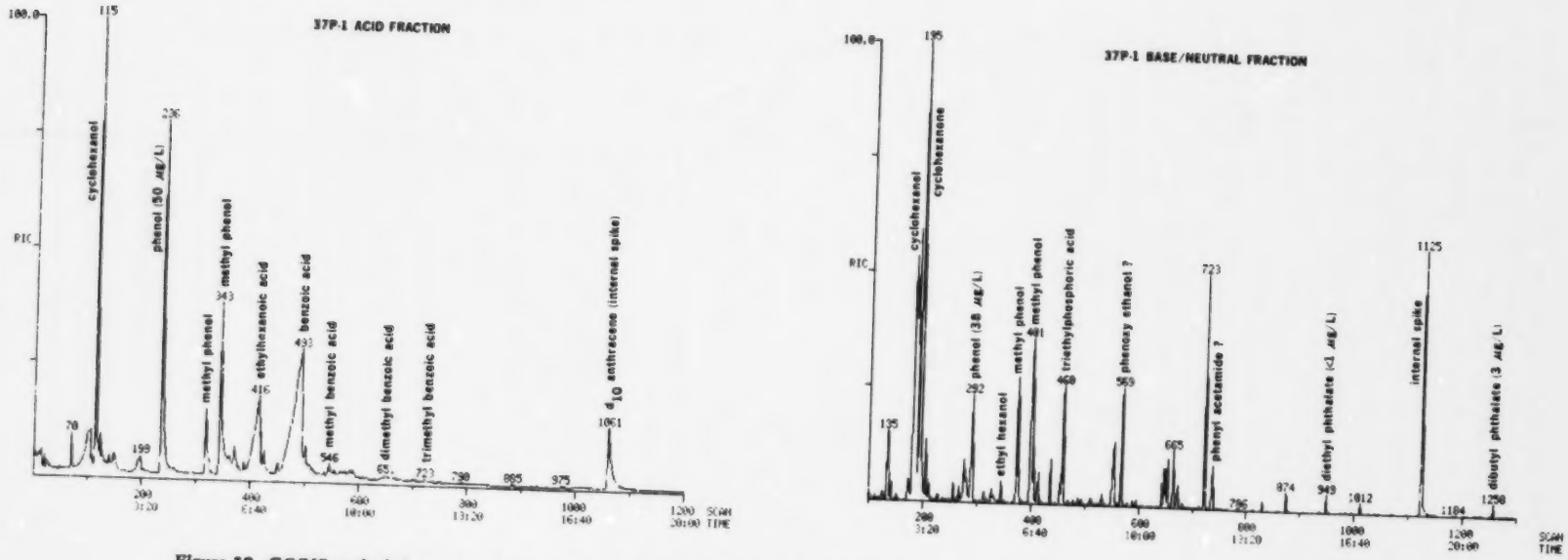

Figure 39. GC/MS analysis (reconstructed ion current chromatograms) of extracted organic compounds (acid and base/neutral fractions) from 37P-1.

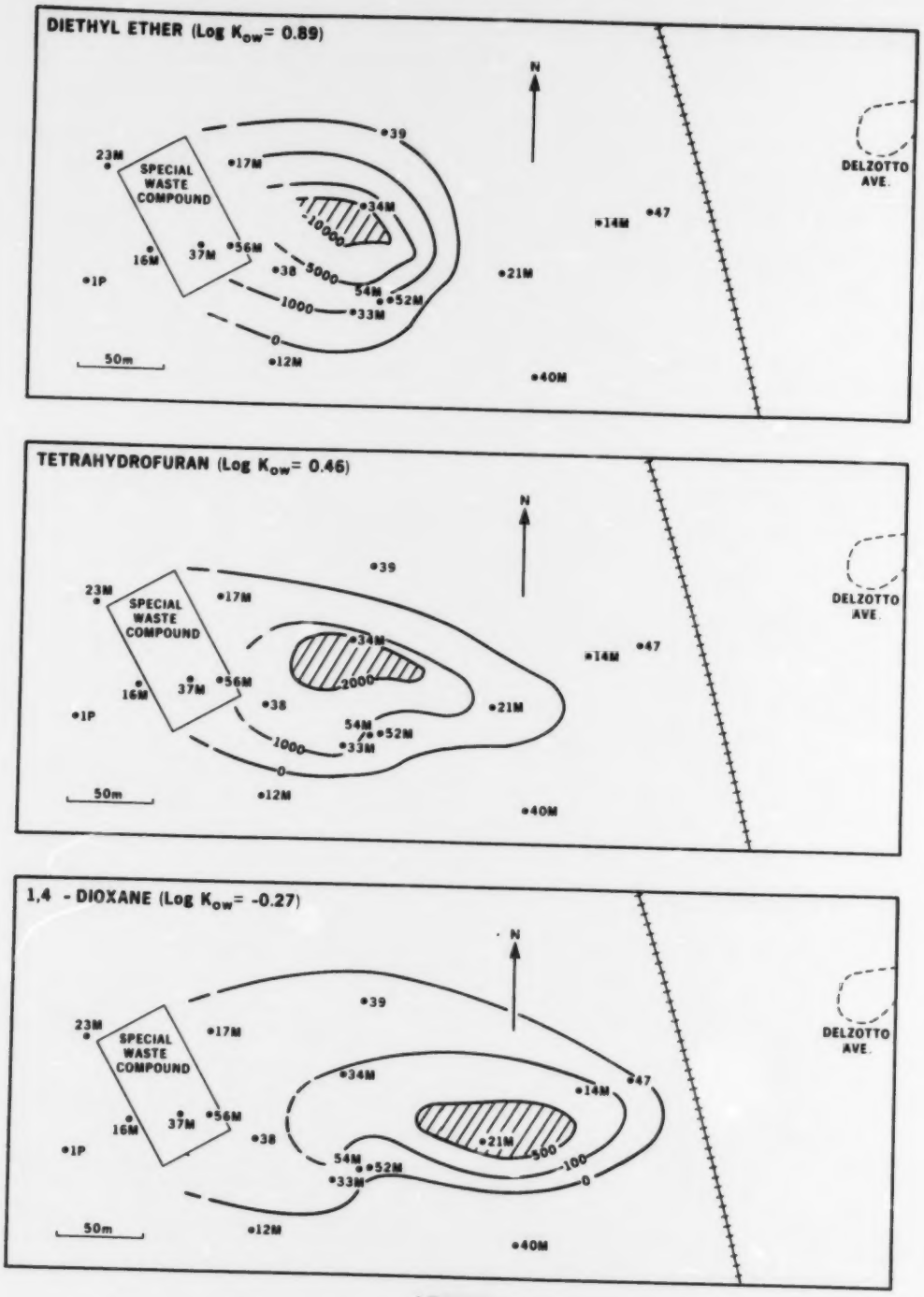

Figure 40. Plan views of plumes for six organic contaminants arranged in order of K_{ow} values.

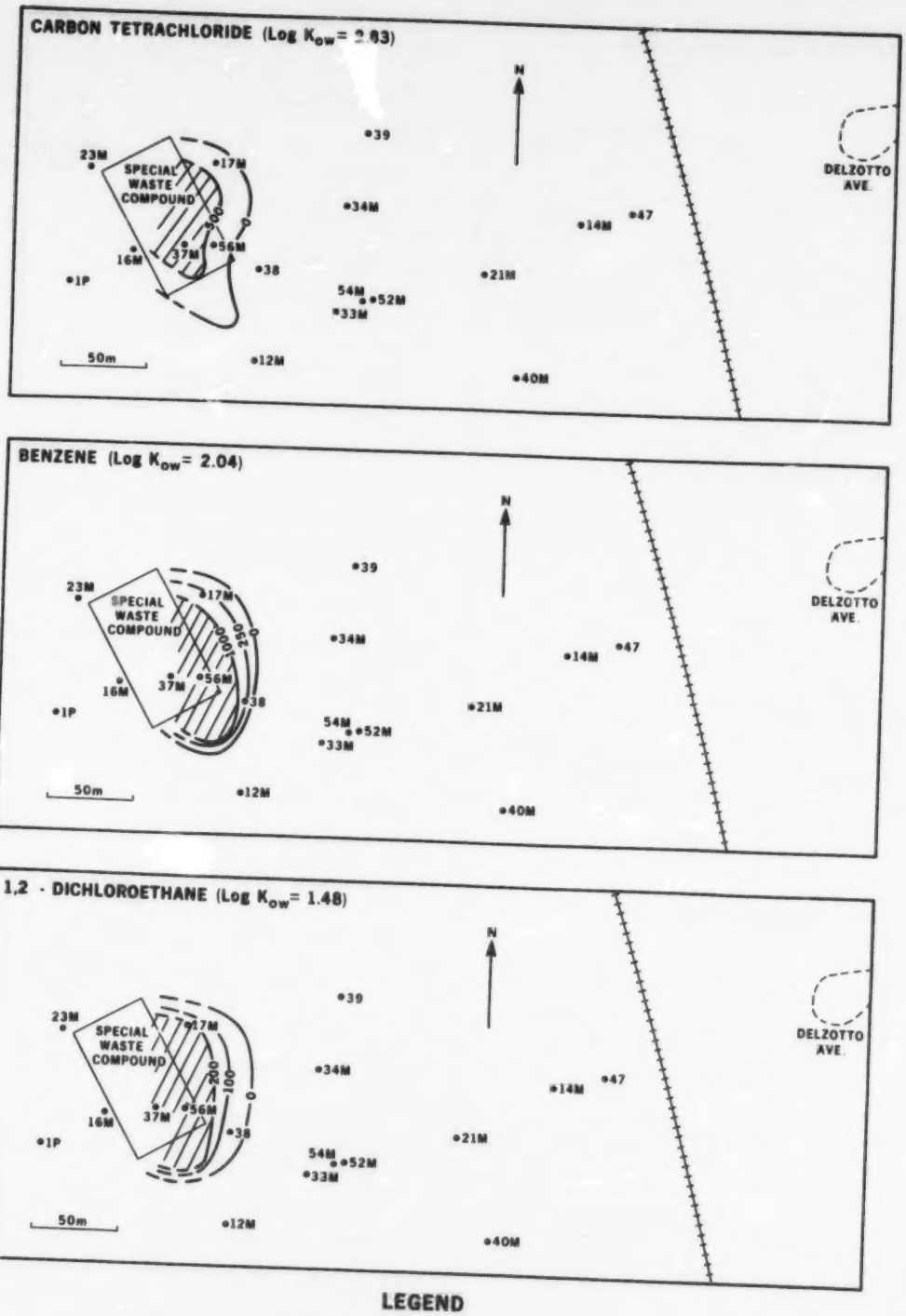

Figure 40. Continued.

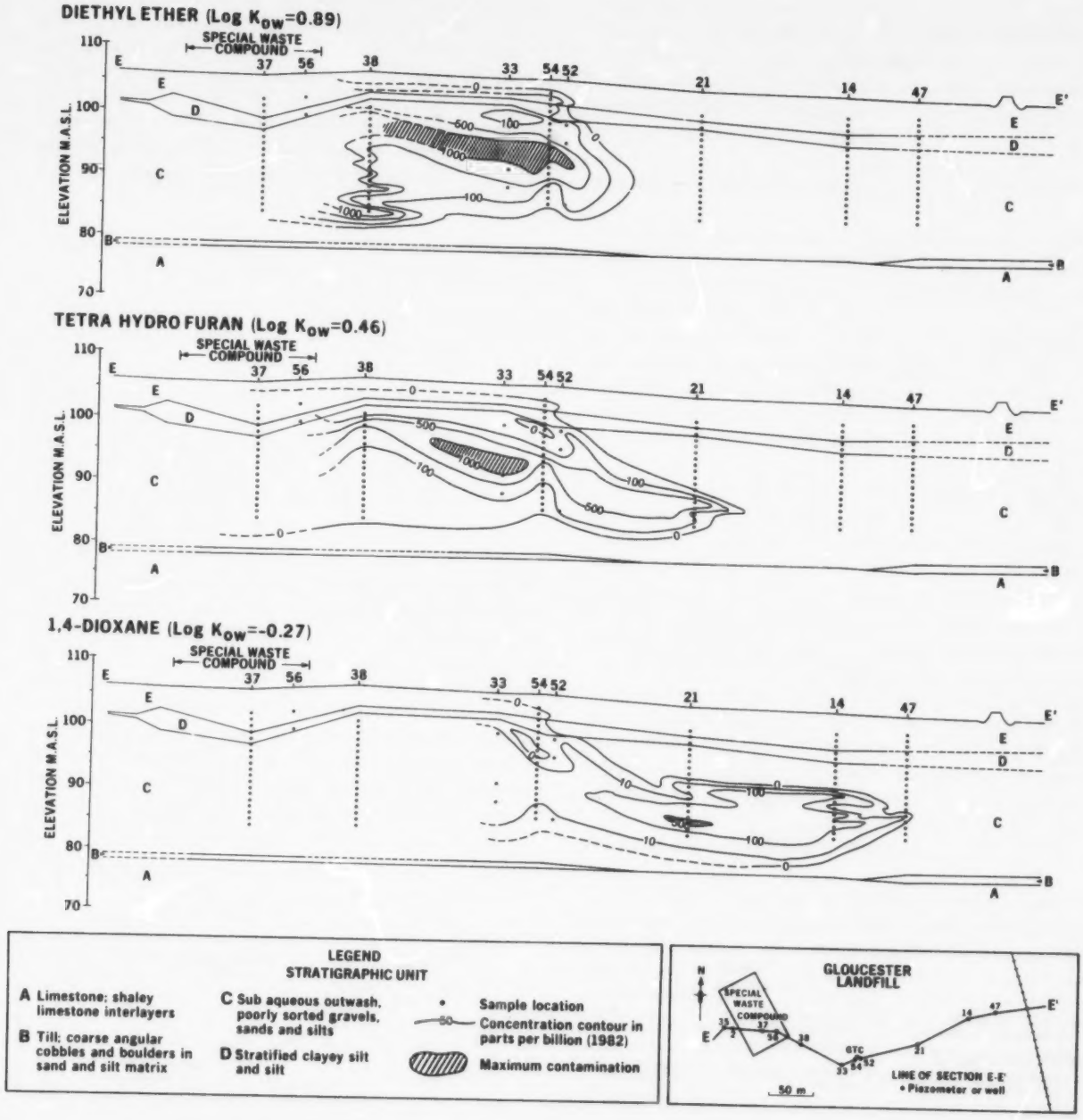

Figure 41. Cross-sectional views of plumes for six organic contaminants arranged in order of K_{ow} values.

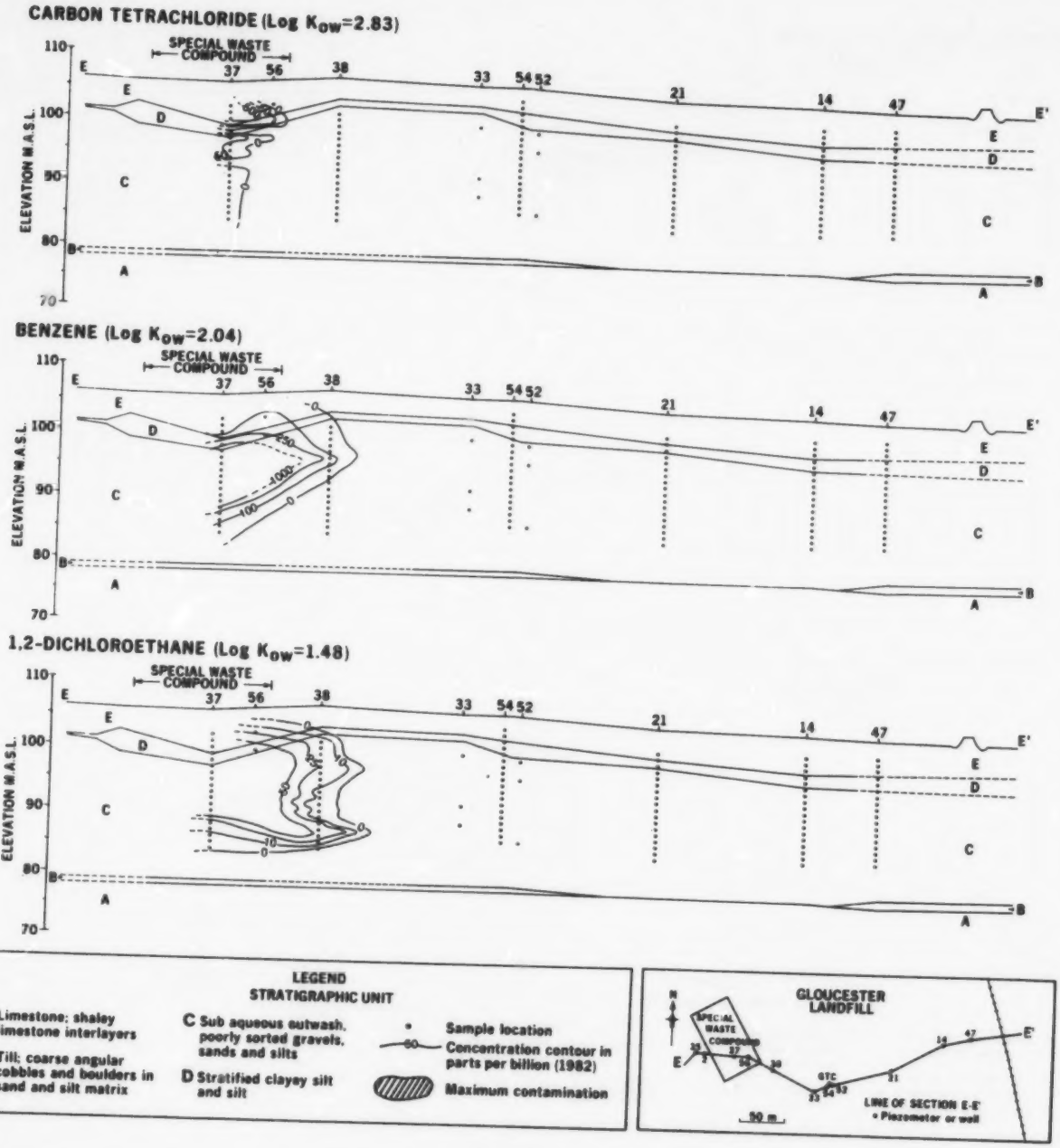

Figure 41. Continued.

By measuring the length of the chloride and organic plumes (assuming location 37 represents the source and that the front of a plume is 0.1 of the maximum concentration) in Figures 34 and 41, values of R_f were calculated for each organic compound (Table 17). Also listed in Table 17 are the average R_f values for diethyl ether, tetrahydrofuran and 1,4-dioxane determined independently during the GTC purge well test. These values are very similar to those derived on the basis of plume lengths. Also included in Table 17 are predicted values from Equation 2.6.

Table 17. Retardation Factors Based on Plume Lengths (R_{f1}), the GTC Purge Well Evaluation Test (R_{f2}) and Equation 2.6, Assuming Either f_{oc} = 0.0035 (R_{f3}) or f_{oc} = 0.01 (R_{f4})

Compound	Log K_{ow}	R_{f1}	R_{f2}	R_{f3}	R_{f4}
1,4-Dioxane	−0.24	1.6	1.4	1.0	1.1
Tetrahydrofuran	0.46	2.2	2.2	1.1	1.4
Diethyl ether	0.89	3.3	3.0	1.3	1.8
1,2-Dichloroethane	1.48	7.6	N.P.	1.8	3.1
Benzene	2.04	8.8	N.P.	2.9	6.4
Carbon tetrachloride	2.83	23.	N.P.	8.0	21.

N.P. — Not present in the test section of the aquifer.

In Figure 42, log (R_f-1) values have been plotted against corresponding log K_{ow} values. The distribution of data points indicates a linear relationship ($r > 0.99$) with a slope of 0.50, which although lower than 0.72 (Equations 2.6 and 5.2), is within the range of values measured by Schwarzenbach and Westall (1981). Since the sediments from the lower aquifer at the Gloucester site contain more than 0.1% organic carbon, the slope of 0.50 suggests that the lipophilicity of the organic matter (Equation 2.5) is small when compared with that of the Swiss sediments studied by Schwarzenbach and Westall. This difference in lipophilicity may reflect the origin of the sedimentary organic matter. A significant fraction of the organic matter in the aquifer sediments at Gloucester was derived from the Paleozoic carbonate bedrock rather than from organisms which lived at the time of sedimentation. Consequently, the organic matter in the Gloucester sediments has probably been modified to a much greater extent by diagenetic processes.

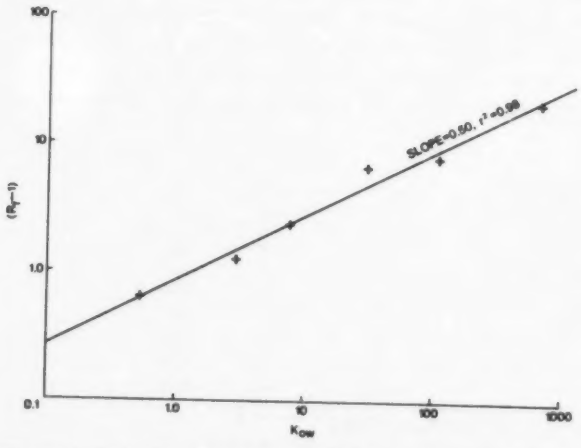

Figure 42. Estimation of the lipophilicity constant ("a" in Equation 2.5) of Schwarzenbach and Westall (1981).

Nonvolatile Organic Contaminants

Several GC/MS analyses have been made to detect very toxic nonvolatile chlorinated compounds suspected to be present in ground waters beneath the Special Waste Compound. Generally, these analyses have shown these nonvolatile compounds are present only in small quantities, if at all (e.g. Fig. 39). Mann Testing Laboratories (1983) did show that two phenoxy herbicides, silvex and 2,4-D, were present in quantities of ~20 ppb in 36W; they detected no pentachlorophenol or DDT.

A variety of polychlorinated dioxins and furans have been detected in a 10-L sample from 37M-19; the congener profile for this sample is shown in Figure 43. This pattern is similar to those observed by Czuczwa and Hites (1984), who attributed it to the combustion of chlorinated organic products, e.g., the chlorinated solvents combusted in the trenches in the Special Waste Compound between 1969 and 1980. The pattern is distinguished by relatively large concentrations of octachlorodibenzodioxins (OCDD) and heptachlorodibenzofurans (HpCDF) and no tetrachlorodibenzodioxin (TCDD). All told there were 59 ppq of polychlorinated dioxins and 43 ppq of polychlorinated furans in this sample. No PCBs were detected.

Transport and Transformation Processes

Although it is apparent that sorption has caused the chromatographic dispersion of the organic contaminants which have entered the outwash aquifer, the role of other physical and chemical processes in affecting the migration of the contaminants is less clear. Of particular interest are (1) the likelihood of two-phase flow involving the simultaneous transport of both aqueous and organic phases and (2) the transformation of the organic contaminants (Chapter 2).

To date, no unequivocal evidence of two-phase flow has been observed. Most significantly there has been no direct observation of a separate organic phase in cores of sediments recovered beneath the Special Waste Compound. Furthermore, maximum concentrations of all identified organic compounds in samples withdrawn from multilevel samplers have been well below their solubility limits

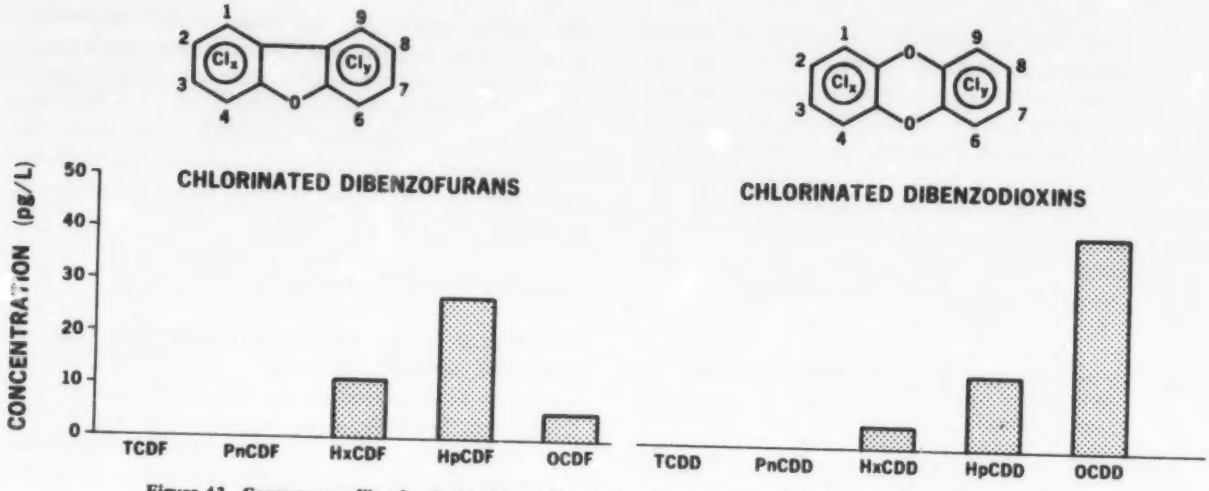

Figure 43. Congener profile of polychlorinated dioxins and furans in a sample of ground water from 37M-19.

(Table 3), indicating the improbability of two-phase flow. For example, the highest concentrations of chloroform or benzene measured at 16M during the period 1980-1983 were <0.1% of their aqueous solubilities; the migration of these plumes beneath the Compound is shown in Figure 44. By the time the organic pollutants have reached 67M (Fig. 45) there is evidence of vertical bifurcation in the plume. However, this is not due to density-induced flow as might be expected if two-phase flow were occurring. Both plumes at 67M contain compounds that have densities both greater and less than that of water. The upper plume contains the maximum values of chloroform (density = 1.5), 1,2-dichloroethane (1.25) and acetone (0.8), and the lower plume contains similar maxima for benzene (0.9), trichloroethene (1.5) and toluene (0.9). Presumably, the two plumes shown in Figure 45 must be due to separate disposals following different flow paths beneath the Compound.

It was noted in Chapter 2 that the transformation of organic contaminants in ground-water flow systems is strongly dependent on the redox environment. Dissolved aqueous sulphide measurements were made (Appendix D) to define the redox state of the ground waters because these measurements are known to give an unambiguous description of anoxic conditions (Jackson and Patterson, 1982). The redox conditions in the outwash aquifer beneath and down-gradient of the Special Waste Plume are summarized in Table 18 and are shown in Figure 46; clearly, the outwash aquifer is in a very anoxic state and conditions are conducive to the transformation of one- and two-carbon aliphatic compounds (Table 6).

The recognition of transformation products in ground waters beneath the Special Waste Compound poses some problems in that a complete list of the compounds disposed of and their quantities is not available. However, the presence of substantial amounts of methanol in the highly reduced zone at 37M-9 and of approximately 100 ppb of ethanethiol (i.e. ethylmercaptan) in 36W (Mann Testing Laboratories, 1983) is believed to be due to their formation *in situ*. The methanol is presumably formed by

Table 18. Redox Conditions Beneath and Downgradient of the Special Waste Compound

Monitor	pH	H_2S (mg/L)	E_H(Pt) (V)	pE(S(VI)/S(-II))	E_H(pE) (V)
37M-1	7.77	0.010	0.09	-9.2	-0.55
37M-9	6.97	0.075	0.13	-8.3	-0.49
67M-9	5.78	1.1	-0.04	-7.1	-0.42
14M-1	7.80	0.005	0.24	-9.2	-0.52
14M-5	7.70	0.035	0.04	-9.4	-0.55
21M-9	7.50	0.005	0.14	-8.9	-0.53

Note: H_2S is analytical total H_2S measured by ion-selective electrode; see Gulens (1985).
E_H(Pt) is the measured Pt electrode potential relative to the normal hydrogen electrode of 0.0 V.
pE(S(VI)/S(-II)) is the pE of the sulphate/sulphide couple from WATEQF (Truesdell and Jones, J. Res. U.S. Geol. Surv., 1974, 2(2), pp. 233-248, Equation 46).
E_H(pE) is the E_H value calculated from pE(S(VI)/S(-II)).

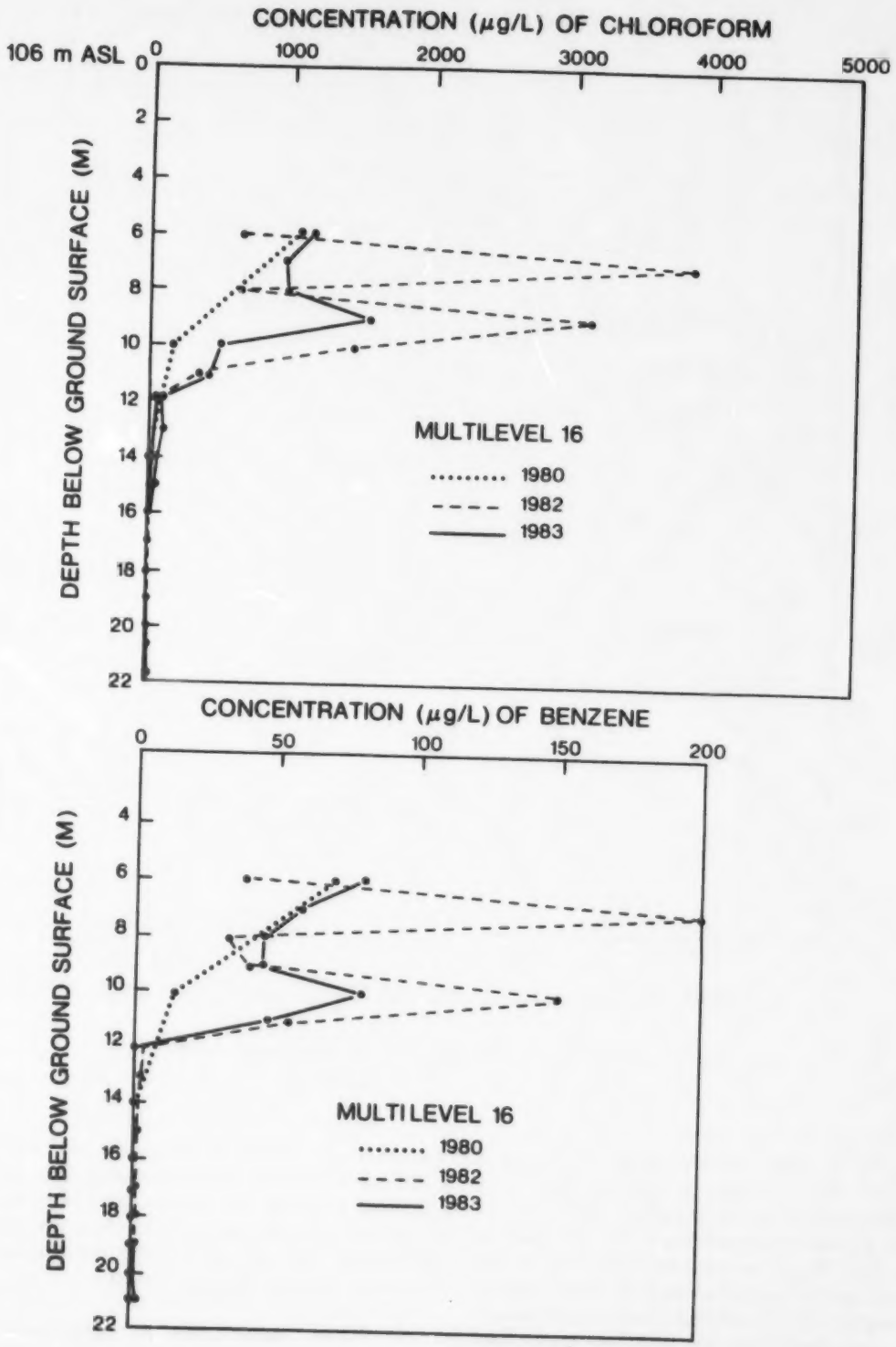

Figure 44. Annual variations in the vertical distribution of volatile organic contaminants at 16M.

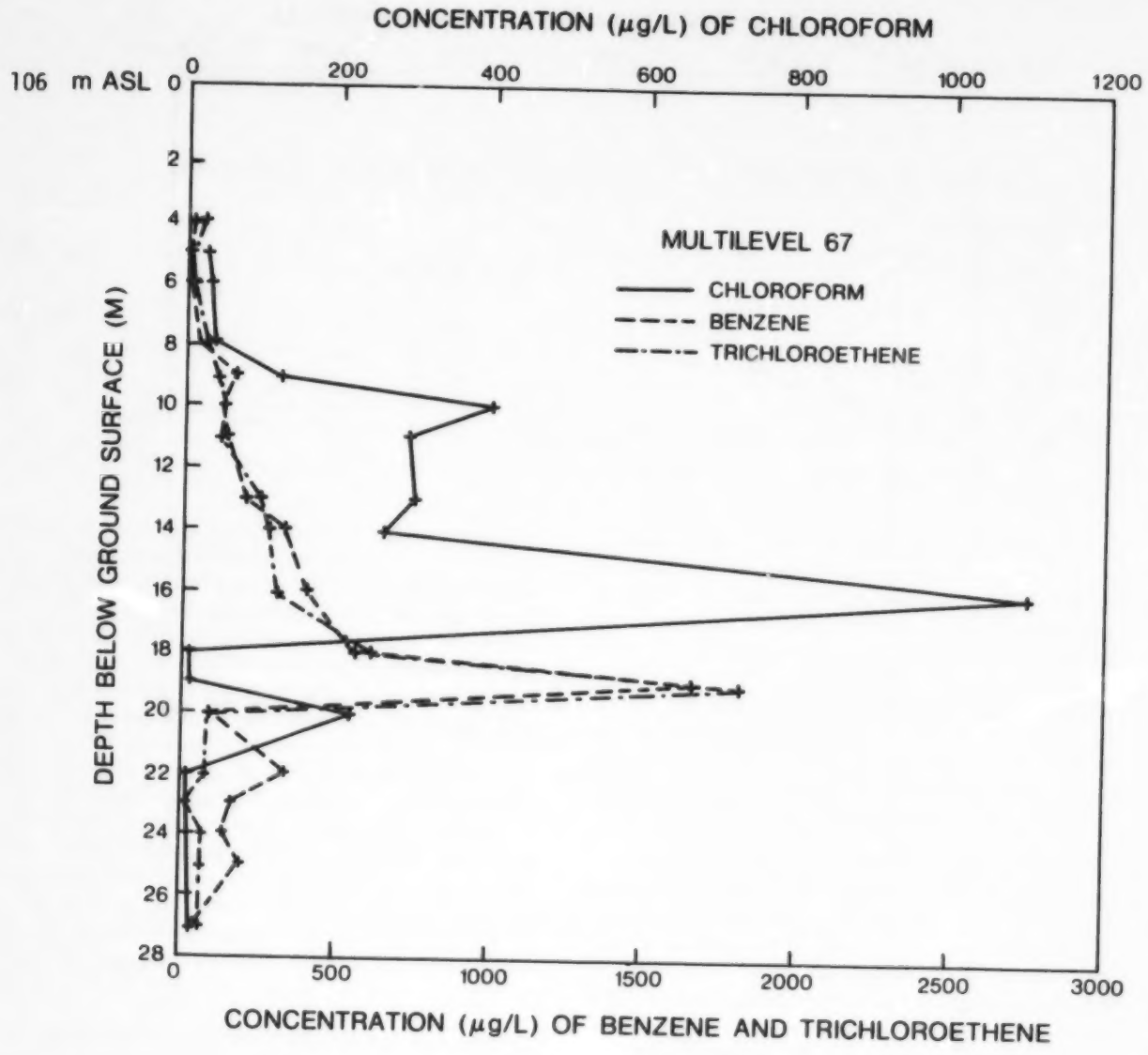

Figure 45. Vertical distribution of volatile organic contaminants at 67M.

the microbially catalyzed transformation of halogenated methanes (either S_N1 with water or S_N2 with OH, Fig. 8). Both Schwarzenbach *et al.* (in press) and McCarty *et al.* (1984) have discussed the possibility of forming thiols or mercaptans (i.e. organic compounds containing an -SH group directly bonded to a carbon atom) in anoxic environments. Schwarzenbach and colleagues have actually demonstrated that halogenated aliphatics have been transformed by nucleophilic substitution reactions to thiols and dialkyl sulphides in anoxic ground waters similar to those at Gloucester (Chapter 2). Furthermore, these workers have stressed the increased importance of such S_N2 reactions (i.e. with -SH groups acting as the nucleophiles) versus hydrolysis reactions at the low temperatures typically found in Canadian aquifers (~10°C).

Further identification of transformation reactions can be attempted by comparing the ratios of precursors to

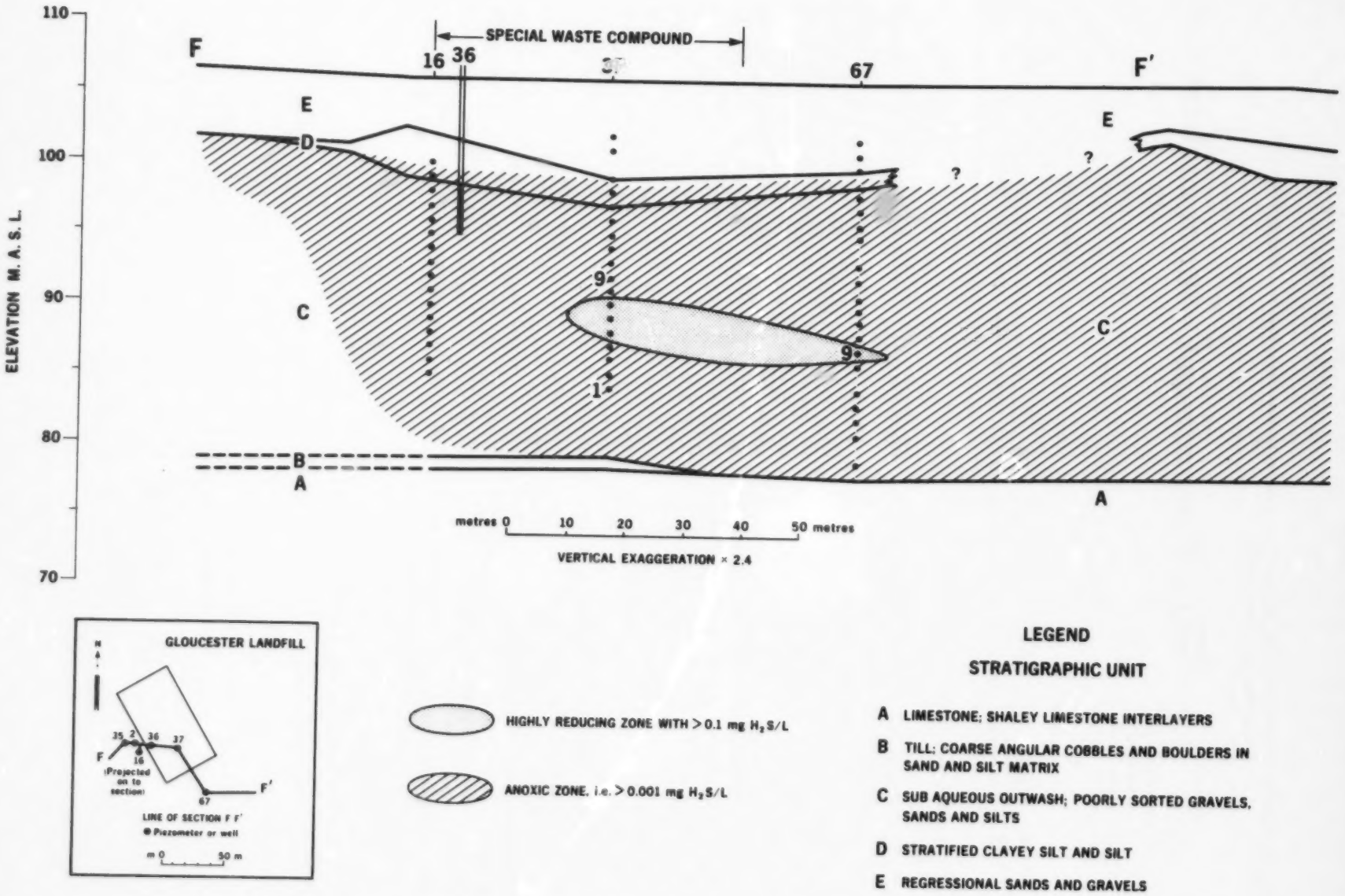

Figure 46. Cross-sectional view through Special Waste Compound showing dissolved sulphide concentrations.

Table 19. Worst-Case Calculation for the Upper Limit on Lifetime Cancer Risk from Drinking a Sample of Ground Water from Monitor 20M

Compound	Concentration (μg/L)		95% Limit × 10^6		Upper limit on lifetime cancer risk
Chloroform	586	×	4.1	=	2402
Carbon tetrachloride	60	×	1.9	=	114
1,2-Dichloroethane	27	×	1	=	27
1,1-Dichloroethane	8	×	150	=	1200
Benzene	100	×	4.4	=	440
				Sum =	4183 × 10^{-6}

Risk calculation for sample from 20M = 1/(4183 × 10^{-6}) = 1 in 240

Note: Concentrations taken from an average of samples from multilevel 20M (Appendix E), representing a worst-case analysis.

products along the path of a plume. In addition to the problem of generally not knowing what was disposed of at a site, two other difficulties arise with this approach. The first concerns the uncertainties arising from the determination of the exact path of a plume beneath a landfill when there may be several overlapping plumes in the same aquifer. The second problem involves the similar effect of sorption and transformation on the value of the ratio with respect to halogenated compounds. Halogen transformation reactions (e.g., Fig. 8) must necessarily result in the decrease of the precursor to product ratio along the path of the plume. It should, however, be noted that the product will be more mobile than its precursor because the effect of losing a halogen atom is to increase the hydrophilic nature of any compound. If, however, all transformation ceased abruptly at some point within the plume, the ratio of precursor to product would still continue to decrease along the path of the plume because of the preferential sorption and retardation of the more halogenated precursor vis-à-vis the less halogenated product.

Health Risks of Drinking Contaminated Ground Water

In Chapter 2, a multistage method was introduced for estimating the potential lifetime cancer risk of drinking contaminated ground water. Table 19 shows concentrations and estimated lifetime cancer risks for a "worst-case" analysis using some Gloucester data. Assuming the risks are additive, the lifetime risk of cancer from lifetime consumption of water from this monitor is approximately 0.0004, or 1 in 240. That is, if all the residents in the Leitrim/Quinn/Delzotto neighbourhood (approximately 200 residents) were to drink this ground water throughout their lifetime, only one additional cancer would likely be caused by the consumption of these five compounds. Because this is a worst-case analysis and this level of contamination has not yet reached the domestic wells in the neighbourhood, it must be concluded that on available evidence, nobody has yet suffered any harm from drinking ground water in this neighbourhood. Nevertheless, a clear threat exists and will have to be removed.

Implications for Site Assessment Studies

The history of the investigations at the Gloucester Landfill and the results of the field and laboratory studies conducted indicate that a reliable assessment of the potential hazard posed by disposal sites can only be determined on the basis of *detailed* hydrogeologic and hydrogeochemical data.

Initially, NHRI became involved at Gloucester in 1979 with the intention of studying the mobility of heavy metals beneath the Special Waste Compound. This was two years after a preliminary site assessment for EPS was undertaken by Gartner, Lee Associates Ltd., Toronto (1978), which laid the hydrogeologic foundations for the later NHRI work. However, it had not been perceived at the time of the Gartner, Lee/EPS study that organic pollutants in ground water should be measured. It was not until early 1981, when 13 samples of ground water from the water table aquifer were analyzed by the Ontario Ministry of the Environment, Rexdale, that the extent of the organic pollution was appreciated. Later in 1981, it became apparent from NHRI's work that a separate plume of volatile organic chemicals was migrating in the lower aquifer from the southern end of the Special Waste Compound and had advanced at least as far as location 21M (e.g. see Fig. 15, April 1982, Annual Report). Detailed mapping of this plume in 1982 and 1983 and the associated hydrogeological work conducted by NHRI and GTC confirmed the extent and degree of contamination. Thus, only after several years of study was the magnitude of the problem recognized.

The results of the first phase of NHRI's activities at the Gloucester Landfill suggest that an assessment of a site suspected of causing ground-water contamination should include the following components:

(1) A detailed hydrostratigraphic evaluation

(2) Substantial field testing to determine the hydraulic properties of the aquifer materials

(3) Determination of the ground-water flow pattern around the site, and in great detail in areas where hazardous waste disposals occurred

(4) *In situ* measurement of ground-water velocities

(5) Detailed chemical sampling with emphasis on those contaminants suspected of being disposed of at the site, e.g. heavy metals, toxic anions (e.g. arsenic) and organic solvents.

The costs of such an assessment would be in excess of $60 000 per site and considerably more should the principal subsurface pathway be in fractured rock. The failure to commit such funds to the task of site assessment may well cause a site to be pronounced uncontaminated when, in fact, serious ground-water pollution could be occurring. That is, the site assessment must be conducted in such a manner to have an acceptably low probability of failing to identify a hazardous situation.

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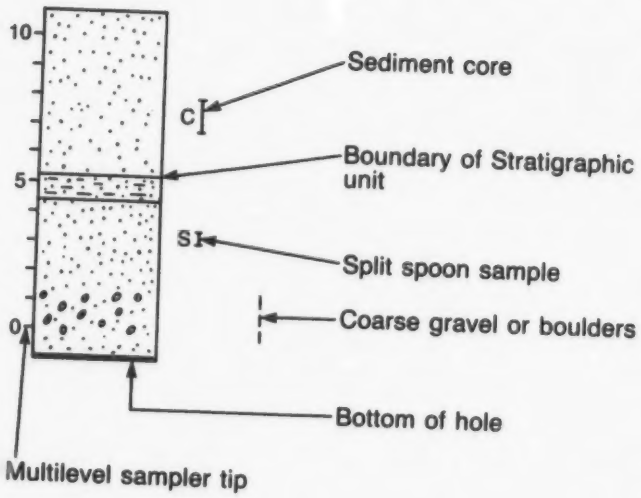
Appendix A
Well Logs, 1983

KEY



STRATIGRAPHIC UNITS

E Regressional sands and gravels

D Stratified clayey silt and silt

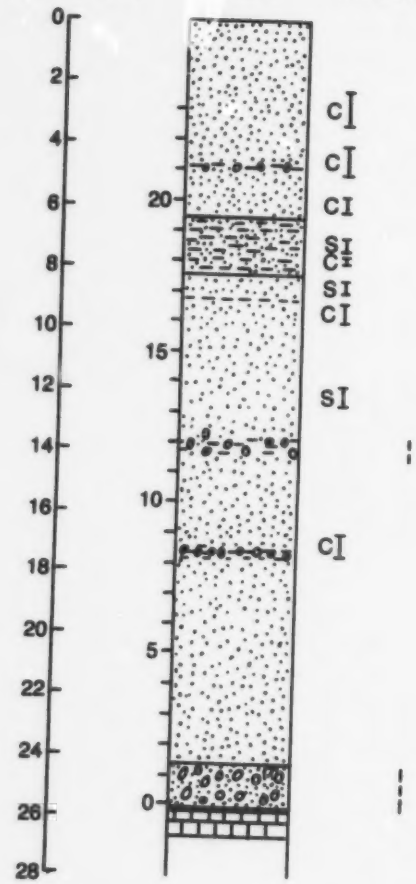
C Sub aqueous outwash; poorly sorted gravels, sands and silts

B Till; coarse angular cobbles and boulders in a sand and silt matrix

A Limestone; shaley limestone interlayers

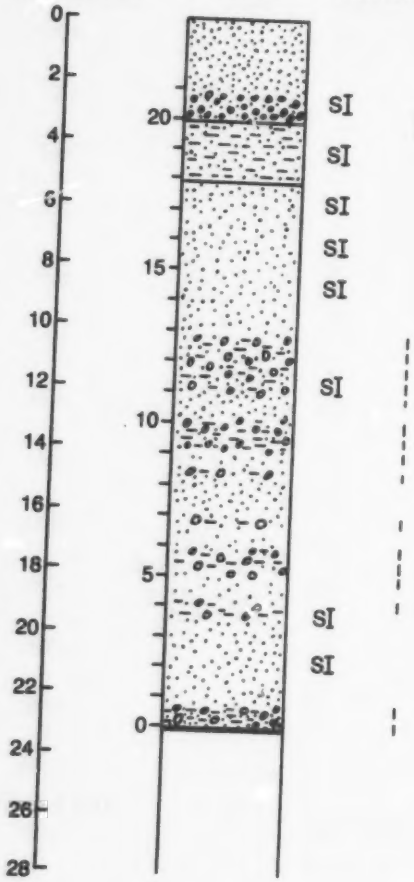
a

STRATIGRAPHIC LOG: MULTILEVEL LOCATION - 82


DEPTH (M)
LOG
SAMPLE TYPE AND DRILLING INDICATIONS
SEDIMENT DESCRIPTION
DESIGNATED UNIT
0
2
4
6
8
10
12
14
16
18
20
22
24
26
28
20
15
10
5
0
CI
CI
CI
SI
CI
SI
CI
SI
CI
- Stratified brown medium and fine sand; some silt laminae and poorly sorted gravel layers at bottom of core
- Stratified fine sands; colour changes from brown to gray 20 cm from bottom of core; ripple drift cross lamination apparent
- Gray fine sand
- Stratified gray silt, fine silt and very fine silt (clay?); finer units laminated on scale of a few mm.
- Gray silt and very fine sand; graded sequences on scale of 0.5 cm
- Gray fine sand; silt (clay?) layers up to 3 cm thick near top of core; fragments of silt (clay?) layers and contorted bedding indicates deformation due to mass flow or dewatering
- Gray fine sand
- Graded sequences up to 25 cm in thickness; in each sequence sediments grade from poorly sorted gray gravel or coarse sand up to gray fine sand
- Dense coarse gravel; large boulders
Bedrock (refusal)
E
D
C
B
A

b

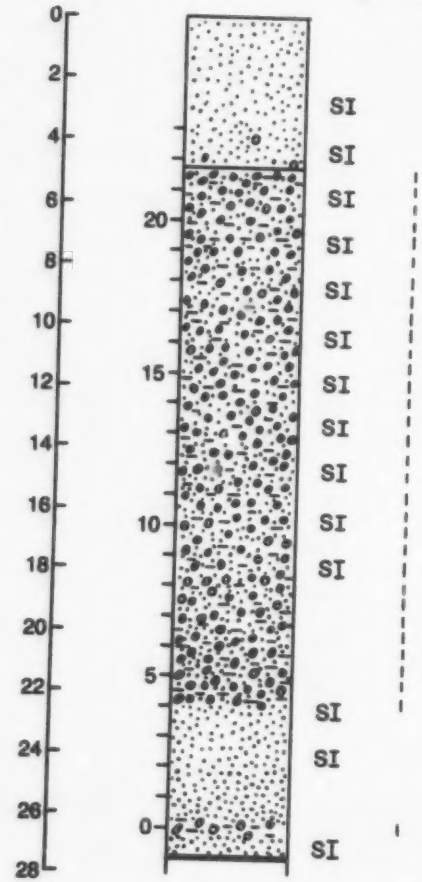
STRATIGRAPHIC LOG: MULTILEVEL LOCATION - 83



SAMPLE TYPE AND DRILLING INDICATIONS	SEDIMENT DESCRIPTION	DESIGNATED UNIT
		—
SI	- Brown silt and fine silt; thin laminae with small pebbles	E
SI	- Stratified gray silt, fine silt, and very fine silt (clay?); minor fine sand lenses and laminae	—
SI	- Gray medium-fine sand	D
SI	- Gray medium and coarse sand	—
SI	- Gray medium and coarse sand	
SI	- Gray, poorly sorted gravel; rounded limestone and shaley limestone pebbles up to 2 cm in diameter	C
SI	- Gray, poorly sorted gravel; rounded limestone and shaley limestone pebbles up to 2 cm in diameter	
SI	- Gray fine-medium sand	
	Boulders (refusal)	

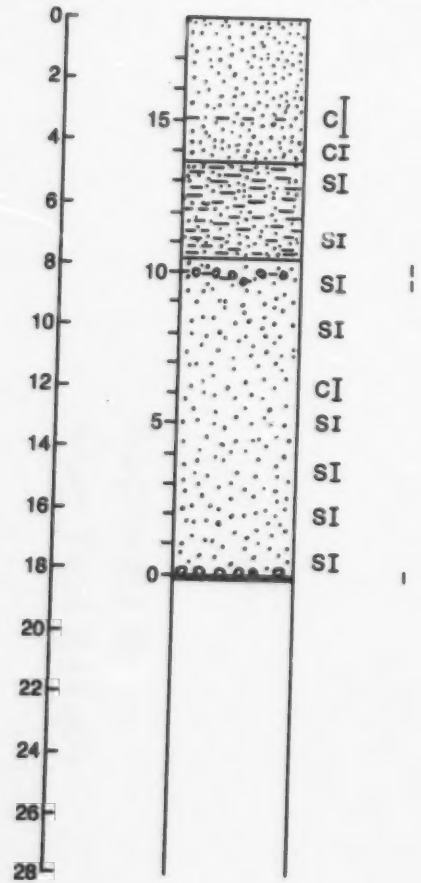
C

STRATIGRAPHIC LOG: MULTILEVEL LOCATION - 84


DEPTH (M)
LOG
SAMPLE TYPE AND DRILLING INDICATIONS
SEDIMENT DESCRIPTION
DESIGNATED UNIT
0
2
4
6
8
10
12
14
16
18
20
22
24
26
28
20
15
10
5
0
SI
- Gray brown fine-medium sand
E
- Brown medium sand; scattered small pebbles
- Dense coarse gravel; angular lithic fragments up to 2 cm in diameter; brown fine-grained matrix
- Brown, poorly sorted gravel; clasts to 3 cm in diameter
- Brown, poorly sorted gravel; clasts to 2 cm in diameter; some crystalline clasts in addition to limestone and shaley limestone
- Gray, poorly sorted gravel; clasts to 2 cm in diameter; some crystalline clasts in addition to limestone and shaley limestone
- Gray, poorly sorted gravel; clasts of limestone and shaley limestone up to 5 cm in diameter
- Gray, poorly sorted gravel; limestone clasts up to 5 cm in diameter
C
- Gray, poorly sorted gravel; clasts to 1 cm in diameter; some crystalline clasts in addition to limestone and shaley limestone
- Gray, poorly sorted gravel; clasts to 2 cm in diameter
- Gray gravel; limestone and shaley limestone clasts up to 3 cm in diameter
- Gray fine and very fine sand
- Gray very fine-fine sand
- Gray fine sand
Auger stuck; drilling terminated

d

STRATIGRAPHIC LOG: MULTILEVEL LOCATION - 85



SAMPLE TYPE AND DRILLING INDICATIONS	SEDIMENT DESCRIPTION	DESIGNATED UNIT
		—
C	- Brown fine-medium sand grading downward into gray fine and medium sand; thin (1 cm) very fine silt (clay?) lamina 30 cm from bottom of core	E
CI	- Gray fine medium sand	
SI	- Weakly laminated gray very fine silt (clay?), fine silt, silt and very fine sand	—
SI	- Weakly laminated gray fine silt and very fine silt; minor lenses of fine sand	D
SI	- Gray fine-very fine sand interstratified with grey poorly sorted gravel with clasts to 4 cm in diameter	—
SI	- Gray silt-very fine sand	
CI	- Gray medium sand with silt and fine sand interlaminations; some sequences graded; some evidence of soft sediment deformation	
SI	- Gray fine sand	C
SI	- Gray very fine sand	
SI	- Gray very fine sand	
SI	- Gray very fine sand Refusal (boulders)	

Appendix B
Slug Test Results

Slug Test Results

Table B-1. Slug Test Results

Piezometer	Hydraulic conductivity (m/s)	Unit
25	2.1×10^{-5}	D
	6.3×10^{-5}	
	6.3×10^{-5}	
	4.1×10^{-5}	
	4.3×10^{-5}	
26	7.0×10^{-6}	D
27-1	5.4×10^{-6}	E
	7.1×10^{-6}	
30-1	1.4×10^{-4}	C
	9.7×10^{-5}	
30-3	4.3×10^{-5}	C
	5.2×10^{-5}	
	7.6×10^{-5}	
	2.8×10^{-6}	
30-5	3.0×10^{-5}	C
	2.9×10^{-5}	
	2.8×10^{-5}	
30-6	9.6×10^{-6}	E
	2.6×10^{-6}	
	4.7×10^{-7}	
30-7	5.9×10^{-6}	E
31-1	5.4×10^{-5}	C
	3.1×10^{-5}	
	4.6×10^{-5}	
	3.2×10^{-5}	
31.2	5.4×10^{-4}	C
	2.1×10^{-4}	
	2.7×10^{-4}	
	1.8×10^{-4}	
31-3	8.9×10^{-5}	C
	6.7×10^{-5}	
	7.6×10^{-5}	
	6.5×10^{-5}	
31-4	6.5×10^{-7}	D
31-5	8.9×10^{-6}	D
	9.3×10^{-6}	
	9.5×10^{-6}	
31-6	4.5×10^{-6}	E
	8.0×10^{-7}	
32-1	7.1×10^{-5}	C
	6.3×10^{-5}	
	7.4×10^{-5}	
	6.3×10^{-5}	
32-3	3.1×10^{-6}	C
	1.6×10^{-4}	
	3.1×10^{-4}	

Table B-1. Continued

Piezometer	Hydraulic conductivity (m/s)	Unit
32-4	6.2×10^{-5}	C
	5.7×10^{-5}	
	5.4×10^{-5}	
	6.3×10^{-5}	
	6.0×10^{-5}	
32-5	9.8×10^{-6}	D
	1.6×10^{-5}	
	2.2×10^{-5}	
32-6	3.6×10^{-5}	E
	2.3×10^{-5}	
	3.3×10^{-5}	
	9.4×10^{-6}	
34-2	2.0×10^{-5}	C
	1.7×10^{-5}	
	2.0×10^{-5}	
	1.8×10^{-5}	
34-3	1.8×10^{-4}	C
	1.5×10^{-4}	
	1.7×10^{-4}	
34-4	3.6×10^{-5}	C
	3.8×10^{-5}	
	3.6×10^{-5}	
	3.8×10^{-5}	
34-5	1.7×10^{-4}	C
	1.5×10^{-4}	
	2.0×10^{-4}	
37-2	5.7×10^{-6}	D
	2.7×10^{-6}	
38-1	5.1×10^{-4}	C
	3.1×10^{-4}	
	1.3×10^{-4}	
45-1	1.7×10^{-6}	D
45-2	1.0×10^{-6}	D
45-3	2.9×10^{-5}	E
	2.7×10^{-5}	
	3.1×10^{-5}	
46-1	1.2×10^{-4}	C
	1.2×10^{-4}	
	1.2×10^{-4}	
46-2	3.1×10^{-5}	D
	2.8×10^{-5}	
	3.8×10^{-5}	
47-1	8.1×10^{-5}	C
	8.7×10^{-5}	
	8.0×10^{-5}	
	7.5×10^{-5}	
47-2	6.9×10^{-6}	D

Table B-1. Continued

Piezometer	Hydraulic conductivity (m/s)	Unit
48-1	3.8×10^{-5}	C
	3.6×10^{-5}	
	1.3×10^{-5}	
	2.9×10^{-5}	
48-2	1.2×10^{-4}	C
	1.1×10^{-4}	
	7.9×10^{-5}	
51-1	1.1×10^{-4}	C
	1.2×10^{-4}	
	3.3×10^{-4}	
	1.1×10^{-4}	
55-1	1.2×10^{-4}	C
	1.3×10^{-4}	
	1.3×10^{-4}	
	1.6×10^{-4}	
	8.7×10^{-5}	

Appendix C
Pump Test Results

Pump Test Results

Table C-1. Pump Test Results (well 49W)

Piezometer	Transmissivity (m^2/s)*				Mean T	Mean S	Mean K (m/s)
	Theis†	Jacob†	Theis‡	Jacob‡			
12-1				7.0×10^{-3} 8.2×10^{-3}	7.6×10^{-3}	2.5×10^{-3}	3.8×10^{-4}
31-1			8.0×10^{-3}	9.2×10^{-3}	8.6×10^{-3}	4.3×10^{-3}	4.3×10^{-4}
31-2			1.1×10^{-2}	8.0×10^{-3}	9.5×10^{-3}	2.3×10^{-3}	4.8×10^{-4}
33-1	8.5×10^{-3}	1.0×10^{-2}	7.5×10^{-3} 6.3×10^{-3}	1.0×10^{-2}	8.3×10^{-3}	3.5×10^{-4}	4.2×10^{-4}
33-2	1.1×10^{-2}	1.1×10^{-2}	5.8×10^{-3} 6.5×10^{-3}	1.1×10^{-2}	8.7×10^{-3}	2.5×10^{-4}	4.4×10^{-4}
33-3			7.7×10^{-3} 5.6×10^{-3}	9.2×10^{-3}	7.3×10^{-3}	3.1×10^{-3}	3.7×10^{-4}
33-4			4.7×10^{-3}	6.7×10^{-3}	5.6×10^{-3}	8.5×10^{-3}	2.8×10^{-4}
34-2	5.2×10^{-3}	1.1×10^{-2}	1.1×10^{-2}	8.7×10^{-3}	8.6×10^{-3}	7.1×10^{-4}	4.3×10^{-4}
34-3	7.9×10^{-3}	9.2×10^{-3}	6.6×10^{-3}		7.8×10^{-3}	3.6×10^{-4}	3.9×10^{-4}
34-4			9.5×10^{-3} 8.5×10^{-3}	1.2×10^{-2}	9.9×10^{-3}	2.5×10^{-4}	5.0×10^{-4}
38-1			6.3×10^{-3}		6.3×10^{-3}	1.8×10^{-4}	3.2×10^{-4}
38-2			5.4×10^{-3}		5.4×10^{-3}	2.5×10^{-3}	2.7×10^{-4}
40-1			3.4×10^{-2}	1.9×10^{-3}	2.5×10^{-2}	2.3×10^{-3}	1.3×10^{-3}
40-2			1.3×10^{-2}		1.3×10^{-2}	3.5×10^{-3}	6.5×10^{-4}
40-3			1.1×10^{-2}	1.6×10^{-2}	1.4×10^{-2}	4.6×10^{-3}	7.0×10^{-4}
40-4			1.2×10^{-2}	2.5×10^{-2}	1.8×10^{-2}	8.0×10^{-3}	9.0×10^{-4}
43W			8.2×10^{-3} 6.4×10^{-3}		7.3×10^{-3}	1.5×10^{-3}	3.7×10^{-4}
46-1	1.3×10^{-2} 6.3×10^{-3}	1.6×10^{-2}	1.3×10^{-2} 1.3×10^{-2}	1.4×10^{-2} 1.4×10^{-2}	1.2×10^{-2}	4.9×10^{-4}	6.0×10^{-4}
46-2	9.9×10^{-3}		1.1×10^{-2}	1.2×10^{-2}	1.1×10^{-2}	3.6×10^{-4}	5.5×10^{-4}
47-1			2.3×10^{-2}	1.6×10^{-2}	1.9×10^{-2}	8.9×10^{-4}	9.5×10^{-4}
47-2				9.7×10^{-3}	9.7×10^{-3}	3.1×10^{-3}	4.9×10^{-4}
51-1	2.6×10^{-3}		1.4×10^{-4}	6.0×10^{-4}	6.0×10^{-4}	5.5×10^{-3}	3.0×10^{-5}
55-1	3.2×10^{-3} 3.0×10^{-3}	6.3×10^{-3}	3.6×10^{-3} 2.6×10^{-3}	1.2×10^{-3}	3.0×10^{-3}	3.2×10^{-4}	1.5×10^{-4}

* Assumes saturated thickness of aquifer = 20 m.
† Short term.
‡ Long term.

Appendix D
Inorganic Chemical Data

Table D-1. Inorganic Analyses, 1980

Site No.	S.C.* (μS/cm)	O_2 (μg/L)	Tit. alk.† (meq/L)	Cl (mg/L)	SO_4 (mg/L)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	Fe (mg/L)	Mn (mg/L)	Cu (μg/L)	Zn (μg/L)	Ag (μg/L)	Pb (μg/L)
16M-1	400		2.79	28	39	35	16.5	1.0	2.5	0.14	0.06	7	5	1	<5
16M-2	450	130		91	37	35	16.0	1.0	2.5	0.20	<0.05	6	<1	1	<5
16M-3	380		2.72	18	32	60	19.0	1.0	2.0	0.18	<0.05	6	<1	<1	<5
16M-4	360		3.16	16	33	55	16.5	1.0	2.0	0.19	<0.05	6	<1	4	<5
16M-5	350	63	2.70	17	31	55	16.5	0.9	2.0	0.19	<0.05	6	200	1	<5
16M-6	385			58	33	60	16.0	0.9	2.0	0.18	0.05	6	4	1	<5
16M-7	340		2.85	14	28	55	16.0	1.0	2.0	0.16	0.05	6	<1	1	<5
16M-8	330		3.11	5.0	24	60	16.0	1.0	2.0	0.17	0.06	5	<1	<1	<5
16M-9	340	237	2.39	30	22	50	16.0	1.0	2.0	0.27	0.12	5	<1	<1	<5
16M-10	320		2.78	14	24	55	16.5	1.0	2.0	0.19	0.20	5	<1	1	<5
16M-11	345		3.52	28	12	60	21.0	0.9	2.5	0.30	0.46	6	<1	<1	<5
16M-12	465		3.62	54	16	75	22.0	1.0	5.0	0.47	0.80	6	1	1	<5
16M-13	700		5.06	60	36	95	27.0	1.6	13.0	–	–	–	–	–	–
16M-14	800		5.40	122	66	110	22.0	3.2	23.0	7.3	2.8	12	5	3	6
16M-15	900	251	5.52	150	77	110	22.5	4.6	25.0	9.6	3.0	9	3	3	<5
16M-16	880		7.08	122	10	130	24.5	3.2	28.0	5.8	3.1	14	2	3	<5
17M-1	265		3.05	17	27	54	14.0	0.6	2.1	0.19	<0.05	6	<1	<1	<5
17M-2	280		3.51	3.2	30	72	24.0	0.7	2.2	0.24	0.12	6	<1	<1	<5
17M-3	295		3.51	4.2	31	58	15.0	0.6	2.2	0.26	0.07	6	1	<1	<5
17M-4	382		3.60	2.9	83	68	17.0	0.8	18.6	0.19	0.54	7	1	1	<5
17M-5	280		3.34	9.4	31	54	15.0	0.6	2.0	0.29	0.05	6	2	<1	<5
17M-6	277		3.35	3.9	34	58	14.0	0.6	2.0	0.19	<0.05	6	<1	<1	<5
17M-7	282		3.30	3.2	31	54	14.0	0.6	2.0	0.17	0.06	7	<1	<1	<5
17M-8	272	19	3.31	5.0	30	56	14.0	0.6	2.3	0.19	0.07	5	1	<1	<5
17M-9	292	10	3.13	13	27	54	15.0	0.6	2.6	0.11	0.08	5	1	<1	<5
17M-10	270	19	3.13	12	18	51	15.0	0.6	2.3	0.08	0.06	6	1	<1	<5
17M-11	283	10	2.57	32	26	52	15.0	0.7	2.3	0.12	0.17	6	1	<1	<5
17M-12	295	15	2.70	18	25	52	14.0	0.7	2.8	0.17	0.28	5	2	<1	<5
17M-13	310	20	1.56	60	60	58	14.0	0.8	3.4	0.20	0.18	8	1	<1	<5
17M-14	400	20	3.56	32	65	67	16.0	0.8	18.6	0.10	0.52	8	<1	1	<5
17M-15			3.44	25	61	60	16.0	0.6	18.3	0.02	0.26	6	1	1	<5
17M-16	375	18	3.29	30	63	62	16.0	0.5	18.1	0.02	0.26	6	<1	1	<5
17M-18	370	2000	3.21	22	73	62	15.0	0.8	18.9	0.01	<0.05	5	1	<1	<5
18M-4	235		2.69	25	38	54	11.5	0.8	2.7	0.19	0.05	5	<1	1	<5
18M-5	200		2.08	21	36	44	10.3	0.8	2.5	0.12	<0.05	4	1	1	<5
18M-6	180		1.69	19	34	37	7.8	0.7	2.0	0.05	<0.05	3	<1	<1	<5
18M-7	160		1.49	12	30	36	7.0	0.6	2.0	0.07	<0.05	3	<1	<1	<5

*Specific conductance.
†Titration alkalinity.

Table D-1. Continued

Site No.	S.C. (µS/cm)	O_2 (µg/L)	Tit. alk. (meq/L)	Cl (mg/L)	SO_4 (mg/L)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	Fe (mg/L)	Mn (mg/L)	Cu (µg/L)	Zn (µg/L)	Ag (µg/L)	Pb (µg/L)
18M-8	140		1.54	8.4	29	33	7.0	0.6	2.0	0.08	0.05	6	<1	<1	<5
18M-9	148		1.57	3.1	29	34	7.5	0.7	2.3	0.05	<0.05	3	<1	<1	<5
18M-10	150		1.57	3.5	28	34	7.3	0.8	2.2	0.05	<0.05	3	1	<1	<5
18M-11	142		0.48	63	27	32	6.8	0.8	2.2	0.06	<0.05	3	<1	<1	<5
18M-12	140		1.56	9.0	27	33	6.8	0.7	2.8	0.07	<0.05	3	<1	<1	<5
18M-13	160		1.84	3.5	26	33	8.3	0.7	3.5	0.04	<0.05	3	<1	<1	<5
18M-14	172		1.67	3.2	30	33	9.3	0.7	4.0	0.05	<0.05	3	<1	<1	<5
18M-15	1050		5.40	220	407	208	51.0	2.1	67.0	0.82	0.07	5	1	1	<5
18M-16	1050		6.25	240	390	210	52.0	1.5	74.0	0.94	0.05	6	<1	1	<5
18M-17	1100		5.10	240	430	210	52.0	1.5	77.0	0.89	0.06	6	<1	1	<5
18M-18	1100		5.38	220	398	196	48.0	1.4	77.0	0.89	0.06	5	1	1	<5
18M-19	1125		6.62	200	402	200	51.0	1.7	82.0	0.97	0.06	5	2	1	<5
18M-20	925		7.20	130	340	170	55.0	1.6	67.0	0.88	0.44	5	<1	1	<5
18M-21	975		9.07	70	235	166	75.0	1.9	38.0	7.8	0.16	1	<1	2	<5
18M-22	900		8.39	54	288	170	57.0	17.5	28.0	14	0.16	1	<1	2	<5
18M-23	1100		10.1	45	118	170	45.0	10.5	170	13	0.42	1	1	3	<5
19M-6	210		1.05	34	10	30	9.0	1.4	3.5	0.06	<0.05	2	<1	<1	<5
19M-7	200		1.41	23	8	25	8.5	1.0	3.5	0.06	<0.05	1	<1	<1	<5
19M-8	380		–	122	2	25	9.5	1.1	4.5	–	–	–	–	–	–
19M-9	270		0.79	80	16	20	10.5	1.2	5.5	0.16	<0.05	2	<1	<1	<5
19M-12	445		2.92	108	16	65	24.5	1.1	14.0	0.05	<0.05	<5	<1	<1	<5
19M-13	445		2.31	165	32	65	21.5	1.1	11.0	0.06	0.05	<5	1	<1	<5
19M-14	750		3.00	165	78	90	26.5	0.9	30.0	1.1	0.09	<5	<1	2	<5
19M-15	800		3.03	185	110	100	27.5	0.9	40.5	2.5	0.10	<5	<1	2	<5
19M-16	910		7.02	150	32	155	48.5	1.8	40.5	9.7	<0.05	6	<1	2	<5
19M-17	1100		13.7	29	14	110	36.5	2.1	39.5	9.7	0.09	4	<1	2	<5
19M-18	810		11.5	21	12	130	20.0	3.0	16.5	72	0.50	7	7	2	<5
19M-19	680		9.44	12	12	90	31.0	3.0	18.0	6.4	0.36	<5	<1	<1	<5
20M-10	470		5.11	46	2	82	30.0	0.9	12.3	12	1.3	5	<1	<1	<5
20M-11	700		4.49	150	2	130	45.0	1.1	16.3	14	1.6	6	1	<1	<5
20M-12	1180		9.48	180	3	235	85.0	1.5	32.5	29	2.3	5	1	1	<5
20M-13	1500		7.26	150	3	335	105	2.1	62.5	15	0.4	5	1	1	<5
20M-14	650		8.75	46	3	111	16.0	0.8	34.0	76	1.2	6	3	<1	<5
20M-15	780		10.3	34	3	96	16.0	7.2	26.5	84	1.6	6	3	<1	<5
20M-16	580		6.79	24	40	70	11.0	2.7	15.0	75	1.4	12	3	<1	<5
20M-17	180		2.25	5.0	16	46	8.0	0.7	1.8	0.82	0.2	3	<1	<1	<5
21M-1	1350		4.88	260	414	195	60.0	2.1	74.0	1.3	0.05	<5	<1	2	6
21M-2	850		6.07	200	140	126	35.0	1.5	48.0	0.87	0.05	<5	<1	2	<5
21M-4	1020		6.16	270	90	153	43.0	1.7	65.0	1.1	<0.05	<5	<1	3	<5
21M-5	720		7.34	170	26	104	32.0	1.5	50.0	1.4	<0.05	<5	<1	2	<5

Table D-1. Continued

Site No.	S.C. (μS/cm)	O_2 (μg/L)	Tit. alk. (meq/L)	Cl (mg/L)	SO_4 (mg/L)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	Fe (mg/L)	Mn (mg/L)	Cu (μg/L)	Zn (μg/L)	Ag (μg/L)	Pb (μg/L)
21M-6	650		5.70	185	6	103	27.0	1.2	14.0	1.0	<0.05	<5	<1	1	<5
21M-7	470		5.64	95	26	102	27.0	0.9	3.4	0.45	<0.05	<5	<1	2	<5
21M-8	620		6.66	46	50	110	32.0	1.1	5.9	0.87	<0.05	<5	<1	2	<5
21M-9	650		7.03	31	90	122	35.0	1.0	13.6	0.59	<0.05	<5	<1	3	<5
21M-10	270		2.92	13	50	56	15.0	0.7	2.7	0.24	<0.05	<5	3	1	<5
21M-11	195		2.34	4.5	23	38	11.0	0.8	2.6	0.08	<0.05	<5	<1	<1	<5
21M-12	195		2.36	5.0	23	39	10.0	0.9	2.4	0.11	<0.05	<5	<1	<1	<5
21M-13	215		2.44	9.2	22	38	18.0	0.8	2.5	0.10	<0.05	<5	<1	<1	<5
21M-14	182		2.33	3.6	16	36	10.0	0.9	2.7	0.12	0.07	<5	<1	<1	<5
21M-15	140		1.21	14	20	26	5.0	0.8	1.5	0.08	0.23	<5	3	<1	<5
21M-16	130		1.24	3.3	14	25	11.0	0.6	1.6	0.03	<0.05	<5	<1	<1	<5
21M-17	140		1.39	3.2	16	26	7.0	0.5	1.6	0.01	<0.05	<5	7	<1	<5
21M-18	125		1.24	3.2	15	26	4.0	0.7	1.5	0.04	0.18	<5	<1	<1	<5
23M-2	220		1.90	35	26	46	13.0	0.7	2.3	0.08	0.05	3	<1	<1	<5
23M-4	225		2.20	26	24	50	14.0	0.9	2.3	0.10	<0.05	7	<1	<1	<5
23M-6	230		1.30	70	30	47	15.0	0.8	2.5	0.11	<0.05	5	<1	<1	<5
23M-8	255		1.75	60	27	52	16.0	0.8	2.0	0.13	<0.05	5	<1	<1	<5
23M-10	235		2.33	25	26	53	14.0	0.8	2.3	0.13	0.09	5	<1	<1	<5
23M-12	218		–	130	15	54	10.0	0.4	1.3	–	–	–	–	–	–
23M-16	165		1.36	21	23	38	9.0	0.7	1.8	0.02	0.08	2	<1	<1	<5

Table D-2. Inorganic Analyses, 1983

Site No.	Depth (m)	S.C.* (µS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (µg/L)	Tit. alk.† (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
5M-1	22.4	205					2.6	2.3	–	52
5M-2	21.4	185					2.4	2.2	–	60
5M-3	20.4	195					2.4	2.4	–	60
5M-4	19.4	135					2.2	4.9	–	17
5M-5	18.4	180					1.9	1.7	–	64
5M-6	17.4	197					1.8	1.4	–	74
5M-7	16.4	190					1.7	0.9	–	48
5M-8	15.4	160					1.8	3.4	–	36
5M-12	11.4	155					1.8	1.0	–	28
5M-14	9.4	135					1.4	0.7	–	28
5M-16	7.4	130					1.4	1.4	–	37
5M-17	6.4	115					1.2	0.6	–	28
5M-18	5.4	115					1.2	0.6	–	27
5M-19	4.4	120					1.2	0.6	–	25
10P-2	2.8							2.5	–	26
12M-6	13.6	155						1.8	–	26
12M-7	12.6	170						0.9	–	24
12M-8	11.6	160						0.5	–	21
12M-9	10.6	165						1.2	–	31
12M-10	9.6	160						0.5	–	24
12M-11	8.6	120						0.8	–	10
12M-12	7.6	100						0.6	–	9.0
12M-13	6.6	100						0.6	–	7.0
12M-14	5.6	125						0.5	–	14
12M-15	4.6	170						0.6	–	19
14M-1	21.3	920						42	–	175
14M-1	21.3					0.005		45	–	130
14M-2	20.3	1145						57	–	205
14M-3	19.3	1105						55	–	165
14M-3	19.3					0.005		46	–	164
14M-4	18.3	610						42	–	40
14M-4	18.3					0.005		45	–	39
14M-5	17.3	780						46	2.5	
14M-5	17.3					0.035		45	2.5	
14M-6	16.3	780						51	–	
14M-6	16.3					0.050		45	–	
14M-7	15.3	780						31	–	25
14M-7	15.3					0.010		45	–	27
14M-8	14.3	445						40	–	
14M-8	14.3					0.004		44	–	9.0
14M-9	13.3	575				0.001		40	–	
14M-10	12.3	565						18	–	
14M-11	11.3	400						–	–	
14M-12	10.3	180						–	–	
14M-13	9.3	175						–	–	10
14M-14	8.3	170						–	–	–
14M-15	7.3	150						–	–	–
14M-16	6.3	140						–	–	–
14M-17	5.3	140						–	–	–
14M-18	4.3	120						–	–	–
14M-19	3.3	115						–	–	–
14M-20	2.3	175						–	–	–
14P-1	12.5		7.73	0.07			2.9	22	–	46
14P-3	28.0		7.30	0.09	0.22		4.7	36	–	80
15M-2	5.3							0.6	–	12
15P	9.0							1.0	–	17

*Specific conductance.
†Titration alkalinity.

Table D-2. Continued

Site No.	Depth (m)	S.C. (µS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (µg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
16M-4	18.2		7.63	0.08			17.4	1.0	–	16
16M-8	14.2		7.94	0.10	0.05		4.6	0.4	–	10
16M-12	10.2		7.68	0.13			5.4	1.6	–	13
16M-16	6.2		7.42	0.20			4.8	2.4	T	16
17M-1	21.3							2.8	–	44
17M-2	20.3							3.1	–	45
17M-3	19.3							2.1	–	35
17M-4	18.3							2.1	–	40
17M-5	17.3							2.2	–	38
17M-6	16.3							2.1	–	33
17M-7	15.3							2.3	–	32
17M-8	14.3							5.4	–	25
17M-9	13.3							5.7	–	23
17M-10	12.3							16	–	26
17M-11	11.3							34	–	34
17M-12	10.3							33	–	53
17M-13	9.3							43	–	96
17M-14	8.3							21	7.0	44
17M-15	7.3							22	15	40
17M-16	6.3							23	15	39
17M-17	5.3							1.8	–	65
17M-18	4.3							1.7	–	63
18M-1	23.3	380						11	–	43
18M-2	22.3	395						15	–	49
18M-3	21.3	390						10	–	39
18M-4	20.3	395						15	–	40
18M-5	19.3	310						12	–	34
18M-6	18.3	265						10	–	35
18M-7	17.3	240						6.0	–	30
18M-8	16.3	210						1.5	–	29
18M-8	16.3					0.014		1.8	–	30
18M-9	15.3	205						0.5	–	21
18M-10	14.3	210						1.4	–	21
18M-10	14.3					0.023		1.1	–	35
18M-11	13.3	200						0.7	–	19
18M-12	12.3	200						1.5	–	33
18M-12	12.3					0.035		0.8	–	29
18M-13	11.3	220						1.0	–	19
18M-14	10.3	220						4.0	–	22
18M-14	10.3					0.046		4.1	–	34
18M-15	9.3	1310						40	–	190
18M-15	9.3					0.006		41	–	211
18M-16	8.3	1410						38	–	210
18M-17	7.3	1395						37	–	250
18M-18	6.3	1277						38	–	300
18M-19	5.3	1211						36	–	230
18M-19	5.3					0.006		41	–	420
18M-20	4.3	1090						26	–	200
18M-21	3.3	995						17	–	60
18M-22	2.3	825						14	–	130
18M-23	1.3	805						15	–	32
19M-2	20.4							1.1	–	22
19M-3	19.4							–	–	15
19M-4	18.4							–	–	13
19M-5	17.4							–	–	15
19M-6	16.4							–	–	14
19M-6	16.4					0.006		1.1	–	20

Table D–2. Continued

Site No.	Depth (m)	S.C. (µS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (µg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
19M-7	15.4							–	–	12
19M-8	14.4							–	–	22
19M-9	13.4							1.0	–	17
19M-10	12.4							45	–	31
19M-11	11.4							45	–	62
19M-12	10.4							30	–	37
19M-13	9.4					0.32		30	–	19
19M-13	9.4							39	–	25
19M-14	8.4							36	–	58
19M-15	7.4							32	–	67
19M-16	6.4					0.009		26	–	38
19M-16	6.4							36	–	45
19M-17	5.4							16	–	55
19M-18	4.4							7.5	–	96
19M-19	3.4							2.6	–	60
21M-1	20.0	1925				0.002		5.4	–	600
21M-1	20.0							44	–	465
21M-2	19.0	1090				0.006		45	–	160
21M-2	19.0							43	–	144
21M-3	18.0	1295						46	–	184
21M-4	17.0	1295				0.006		39	–	163
21M-4	17.0							44	–	152
21M-5	16.0	1120						34	–	138
21M-6	15.0	855				0.020		35	–	97
21M-6	15.0							45	–	12
21M-7	14.0	805						28	–	49
21M-8	13.0	750						20	2.8	63
21M-9	12.0	675				0.005		4.7	–	55
21M-9	12.0							5.8	–	56
21M-10	11.0	350				0.010		5.2	–	24
21M-10	11.0							3.0	–	53
21M-11	10.0	270						–	–	–
21M-12	9.0	265				0.040		–	–	–
21M-12	9.0							0.9	–	20
21M-13	8.0	270						–	–	–
21M-14	7.0	265						–	–	–
21M-15	6.0	190						–	–	–
21M-16	5.0	185						0.5	–	16
21M-17	4.0	180						–	–	–
21M-18	3.0	155						–	–	–
32P-1	22.1	290					3.4	2.0	–	70
32P-2	18.8	255					3.2	1.8	–	50
32P-3	13.6	340					3.0	24	–	125
32P-4	8.9	590					5.9	5.0	–	360
32P-5	7.7	645					6.3	4.6	–	480
32P-6	5.6	630					5.9	3.0	–	400
32P-7	4.0	465					4.6	1.4	–	160
33M-1	21.0	375						2.5	–	36
33M-2	20.0	375						1.8	–	36
33M-3	19.0	505						20	–	51
33M-4	18.0	1105						36	–	50
33M-5	17.0	1190						33	–	88
33M-6	16.0	1105						34	–	76
33M-7	15.0	1040						37	–	100
33M-8	14.0	853						30	–	94
33M-9	13.0	590						21	–	74
33M-10	12.0	400						5.3	–	51

Table D-2. Continued

Site No.	Depth (m)	S.C. (μS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (μg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
33M-11	11.0	315						5.2	–	48
33M-12	10.0	227						2.7	–	31
33M-13	9.0	310						6.9	–	55
33M-14	8.0	235						3.9	–	54
33M-15	7.0	410						8.5	–	75
33M-16	6.0	335						0.9	–	26
33M-17	5.0	340						1.2	–	30
33M-18	4.0	270						1.1	–	14
34M-2	19.8	820						25	–	–
34M-3	18.8	1064						44	–	–
34M-4	17.8	885						44	–	–
34M-5	16.8	885						38	–	–
34M-6	15.8	640						34	–	–
34M-7	14.8	565						29	–	–
34M-8	13.8	490						26	–	–
34M-9	12.8	465						25	–	–
34M-10	11.8	400						18	–	–
34M-11	10.8	455						23	–	–
34M-12	9.8	450						25	–	–
34M-13	8.8	655						–	–	–
34M-14	7.8	955						–	–	–
34M-15	6.8	841						–	–	–
34M-16	5.8	995						–	–	–
34M-17	4.8	1154						–	–	–
34M-18	3.8	1430						–	–	–
34M-19	2.8	2500						–	–	–
35P-1	9.0	–					3.8	0.5	T	24
37M-1	22.0		7.77	0.07	0.66	0.010	4.8	1.2	–	36
37M-1	22.0						4.6	–	–	–
37M-2	21.0		7.68	0.07	0.85		5.9	–	–	–
37M-3	20.0	450	7.59	0.09	1.20		8.0	63	22	63
37M-4	19.0	610	7.10	0.10	0.55	0.030		46	14	58
37M-4	19.0						9.2	74	18	86
37M-5	18.0	765	7.08	0.06	1.80	0.11		45	9.0	47
37M-5	18.0						9.5	72	19	79
37M-6	17.0	725	7.05	0.06	0.28	0.12		45	7.0	46
37M-6	17.0							39	17	80
37M-6	17.0						8.5	82	20	87
37M-7	16.0	800	7.02	0.06	0.11	0.13		45	11	62
37M-7	16.0							40	18	87
37M-7	16.0						13.8	86	19	350
37M-8	15.0	1185	6.95	0.04	0.028	0.083		39	17	400
37M-8	15.0						12.8	60	7.8	220
37M-9	14.0	975	6.97	0.04	0.026	0.075		45	10	155
37M-9	14.0							39	7.0	250
37M-9	14.0						12.7	46	3.5	122
37M-10	13.0	540	6.97	0.11	0.33	0.020		45	6.0	117
37M-10	13.0							40	3.1	120
37M-10	13.0						10.0	36	–	56
37M-11	12.0	555	7.09	0.11	0.31	0.099		26	2.0	63
37M-11	12.0							40	T	61
37M-11	12.0						7.2	17	–	37
37M-12	11.0		7.23	0.14	0.37			24	–	61
37M-12	11.0						7.4	13	–	35
37M-13	10.0		7.30	0.15	0.72			17		39
37M-13	10.0						5.9	10	–	33
37M-14	9.0		7.47	0.09	0.65					

Table D-2. Continued

Site No.	Depth (m)	S.C. (µS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (µg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
37M-14	9.0					0.010		7.6	–	38
37M-14	9.0							12	–	36
37M-15	8.0		7.21	0.13	0.88		8.1	17	2.2	155
37M-16	7.0		7.17	0.20	0.36		10.0	14	–	175
37M-17	6.0		7.13	0.28	0.74		7.4	12	–	160
37M-18	5.0		7.09	0.35	0.86		9.5	12	–	160
37M-19	4.0		7.10	0.32	0.51		10.2	12	–	280
37P-1	16.7	660					6.8	62	12	64
37P-4	34.4							11	–	75
38M-1	22.8	350						5.2	–	11
38M-2	21.8	380						3.1	–	44
38M-3	20.8	380						2.4	–	35
38M-4	19.8	385						8.0	–	20
38M-5	18.8	385						12	–	31
38M-6	17.8	365						8.9	–	11
38M-7	16.8	380						7.6	–	31
38M-8	15.8	355						3.0	–	33
38M-9	14.8	380						5.9	–	33
38M-10	13.8	375						8.9	–	40
38M-11	12.8	400						14	–	35
38M-12	11.8	410						19	–	45
38M-13	10.8	460						16	–	36
38M-14	9.8	435						22	–	34
38M-15	8.8	510						26	–	47
38M-16	7.8	545						7.7	–	29
38M-17	6.8	345						6.7	–	145
38M-18	5.8	720						36	–	–
39M-1	21.9	605						34	–	–
39M-2	20.9	565						29	–	25
39M-3	19.9	435						24	–	16
39M-4	18.9	360						23	–	25
39M-5	17.9	370						12	–	100
39M-6	16.9	415						24	–	210
39M-7	15.9	730						26	–	280
39M-8	14.9	1295						23	–	240
39M-9	13.9	795						8.4	–	195
39M-10	12.9	505						6.7	–	200
39M-11	11.9	580						24	–	210
39M-12	10.9	1025						19	–	190
39M-13	9.9	885						23	–	195
39M-14	8.9	805						24	–	190
39M-15	7.9	790						20	–	190
39M-16	6.9	910						17	–	190
39M-17	5.9	895						14	–	185
39M-18	4.9	820						5.2	–	110
39M-19	3.9	688						4.2	–	32
40M-1	20.8	375						1.8	–	28
40M-2	19.8	330						1.4	–	23
40M-3	18.8	320						1.2	–	23
40M-4	17.8	295						0.7	–	20
40M-5	16.8	270						–	–	16
40M-6	15.8	245						–	–	14
40M-7	14.8	245						–	–	12
40M-8	13.8	225						1.0	–	36
40M-9	12.8	210						3.5	–	31
40M-10	11.8	205						3.0	–	32
40M-11	10.8	200								

Table D-2. Continued

Site No.	Depth (m)	S.C. (μS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (μg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
40M-12	9.8	215						2.8	–	31
40M-13	8.8	215						2.9	–	37
40M-14	7.8	205						2.8	–	34
40M-15	6.8	240						2.7	–	37
40M-16	5.8	315						2.5	–	34
40M-17	4.8	325						2.6	–	37
40M-18	3.8	325						2.7	–	20
44W	10.3							1.5	–	29
44W	10.3							1.7	–	27
45P-1	18.3	145					1.6	0.6	–	23
45P-2	12.2	100					0.7	4.7	–	12
45P-3	6.1	70					0.7	5.0	–	–
47M-1	19.6	820						44	2.4	110
47M-2	18.6	625						44	–	74
47M-3	17.6	855						45	–	100
47M-4	16.6	780						44	3.5	46
47M-5	15.6	665						37	4.0	42
47M-6	14.6	590						34	–	31
47M-7	13.6	590						29	–	74
47M-8	12.6	450						9.7	–	77
47M-9	11.6	250						1.8	–	22
47M-10	10.6	250						1.2	–	21
47M-11	9.6	160						0.9	–	17
47M-12	8.6	180						0.8	–	20
47M-13	7.6	190						1.1	–	22
47M-14	6.6	165						0.8	–	16
47M-15	5.6	145						0.6	–	12
47M-16	4.6	135						0.6	–	10
47M-17	3.6	155						0.9	–	14
47M-18	2.6	135						0.6	–	10
48P-1	19.8	260					3.6	1.5	–	28
48P-2	15.2	200					2.8	3.2	–	11
48P-3	6.0	125					1.7	1.4	–	–
52M-8	16.0							39	2.8	134
52M-9	15.0							36	3.7	135
52M-10	14.0							33	–	127
53M-5	16.3							42	3.0	100
53M-6	15.3							41	4.6	107
53M-7	14.3							39	2.3	118
54M-1	23.0	562						36	–	55
54M-2	22.0	462						24	–	49
54M-3	21.0	429						18	–	45
54M-4	20.0	1184						37	–	
54M-5	19.0	1122						49	–	49
54M-6	18.0	1199						54	–	26
54M-7	17.0	1150						31	–	48
54M-8	16.0	1418						35	–	73
54M-9	15.0	1205						31	–	63
54M-10	14.0	1095						28	–	58
54M-11	13.0	648						20	–	33
54M-12	12.0	393						15	–	15
54M-13	11.0	549						13	–	27
54M-14	10.0	536						17	–	29
54M-15	9.0	518						17	3.0	42
54M-16	8.0	454						2.8	–	40

Table D-2. Continued

Site No.	Depth (m)	S.C. (µS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (µg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
54M-17	7.0	439						0.6	–	16
54M-18	6.0	330						0.7	–	14
54M-19	5.0	297						0.5	–	11
60P-1	5.6						2.1	1.4		10
67M-1	27.0	280						0.6	–	30
67M-3	25.0	300						1.6	–	35
67M-3	25.0		7.73	0.06	0.06		3.7	0.6	–	36
67M-4	24.0	220						0.5	–	28
67M-5	23.0	230						0.7	–	27
67M-6	22.0	225						1.3	–	38
67M-6	22.0					0.005		3.0	–	40
67M-7	21.0	300						1.3	–	38
67M-7	21.0					0.005		2.6	–	36
67M-8	20.0	350						45	16	149
67M-8	20.0							14	–	21
67M-9	19.0	530						9.3	–	41
67M-9	19.0					1.1		21	T	50
67M-9	19.0		5.78	0.05	0.055		7.5	36	–	58
67M-10	18.0	640						45	2.0	78
67M-10	18.0					0.020		42	2.0	110
67M-11	17.0	685						45	2.0	164
67M-11	17.0					0.040		50	–	80
67M-12	16.0	860						45	2.0	177
67M-12	16.0					0.005		70	–	50
67M-14	14.0	805						45	–	–
67M-14	14.0					T				
67M-15	13.0	930						45	–	16
67M-15	13.0					0.010				
67M-17	11.0	635						32	–	160
67M-18	10.0	685						5.0	15	30
67M-19	9.0	495						15	–	205
67M-19	9.0		7.54	0.14	1.0		7.0			
67M-20	8.0	405						2.0	100	30
67M-21	7.0	485						1.8	80	25
67M-22	6.0	425						0.5	20	20
67M-23	5.0	230						3.0	12	15
67M-24	4.0	220								
68M-1	23.7		7.65	0.08	0.105		3.2	17	–	49
68M-7	17.7		7.84	0.05	0.12		3.2	3.7	–	34
68M-12	12.7							<1	–	20
68M-23	1.7		6.90	0.25	0.21		2.6	1.5	–	37
73M-5	3.7	468						2.7	–	60
73M-6	2.7	445						0.8	–	54
74M-1	6.4	735						48	–	83
74M-2	5.4	1116						57	–	187
74M-3	4.4	1238						26	–	101
74M-4	3.4	1337						23	–	137
74M-5	2.4	735						4.0	–	24
74M-6	1.4	647						3.0	–	–
75M-1	6.0	924						52	–	134
75M-2	5.0	733						3.5	–	62
75M-3	4.0	668						1.0	6.9	41
75M-4	3.0	535						1.4	–	50
75M-5	2.0	622						1.6	–	61
75M-6	1.0	336						–	–	10

Table D-2. Continued

Site No.	Depth (m)	S.C. (μS/cm)	pH	E_H (V)	O_2 (mg/L)	S^{2-} (μg/L)	Tit. alk. (meq/L)	Cl (mg/L)	Br (mg/L)	SO_4 (mg/L)
76M-2	5.0	1016						27	–	133
76M-3	4.0	515						1.1	–	59
76M-5	2.0	495						1.2	–	49
83M-1	22.2							40	–	65
83M-3	20.2							>40	–	283
83M-5	18.2							>40	–	269
83M-7	16.2							37	–	86
83M-9	14.2							4.5	–	37
83M-11	12.2							7.4	–	37
84M-0	26.5							–	–	27
84M-2	24.5							–	–	24
84M-4	22.5							–	–	23
84M-5	21.5							–	–	24
84M-5	21.5							2.8	–	25
84M-6	20.5							–	–	31
84M-7	19.5							1.3	–	41
84M-19	7.5							2.8	–	25
85M-0	18.3							>40	–	28
85M-1	17.3							>40	–	40
85M-1	17.3							>40	–	39
85M-2	16.3							>40	–	49
85M-4	14.3							6.4	–	53
85M-6	12.3							1.6	–	24
85M-15	3.3							1.0	–	–
GTCM1-1	25.7							62	–	147
GTCM1-2	24.7							62	–	145
GTCM1-4	23.7							48	–	77
GTCM1-5	22.7							63	–	117
GTCM1-6	21.7							63	–	159
GTCM1-7	20.7							63	–	224
GTCM1-8	19.7							63	–	225
GTCM1-9	18.7							62	–	89
GTCM1-10	17.7							63	–	61
GTCM1-12	16.7							61	–	255
GTCM1-13	15.7							63	–	173
GTCM1-14	14.7							62	–	144
GTCM1-15	13.7							63	–	184
GTCM1-16	12.7							63	–	194
GTCM1-17	11.7							63	–	133
GTCM1-18	10.7							63	–	192
GTCM1-19	9.7							63	–	161
GTCM1-20	8.7							64	–	101
GTCM1-21	7.7							63	–	109
GTCM1-22	6.7							63	–	86
GTCM1-23	5.7							11	–	34
GTCM1-24	4.7							7.0	–	30
GTCM1-25	3.7							7.0	–	22

Appendix E
Organic Chemical Data

Table E-1. Organic Chemical Data Abbreviations

Compound	Abbreviation
Dichloromethane	CH2CL2
Acetone	ACET
1,1-Dichloroethene	11DCEE
1,1-Dichloroethane	11DCEA
Tetrahydrofuran	THF
Trans-1,2-dichloroethene	T12DE
Chloroform	CHCL3
Diethyl ether	DEE
1,2-Dichloroethane	12DCEA
1,1,1-Trichloroethane	111TCEA
1,4-Dioxane	14DNE
Carbon tetrachloride	CCL4
Trichloroethene	TCEY
Benzene and three other compounds	BENZ
Bromoform	CHBR3
Tetrachloroethene	TECE
Toluene	MEBEN
Chlorobenzene	CLBEN
Ethylbenzene	ETBEN

Note:

A – Methanol present.
B – Bromoform resolution questionable.
T – Trace of contaminant found.

11DCEA THF
CHCL3 DEE
111TCEA 14DNE
} Coelution problem with each underlined pair of compounds. Concentration values listed under *most likely* compound.

BENZ – Benzene coelutes with three other compounds.

11DCEE
T12DE
} Identities of these compounds *not* confirmed.

Table E-2. December 1982 GC/MS Confirmation (μg/L)

Site No.	CH2CL2	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEY	BENZ	CHBR3	TECE	MEBEN	ETBEN
16M-16	620	20				12 900		480	1900			1600	580	90	4500	1000	16
17M-16						3 900		240	6		250	250					
19M-16					150	470		110				750	230				
21M-3										120							
34M-4				2000			9700										
37M-9	6050	30	3500			32 000		2700	3500			20	17 000				
37M-19	37					8 500			75		3650	1030		2500	350		
38M-6							1500	25	26							5	
47M-1																	
54M-6				200			700										

Table E-3. September–December 1982 Volatile Organic Analyses (µg/L)

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEY	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
1P-1					22															
1P-3					195															
12M-14																				
14M-1											53									
14M-2						3.4					50									
14M-3						3.3					200									
14M-4						24					100									
14M-5						6.4					165									
14M-6						24					2.0									
14M-7											37									
14M-8						8.5					120									
14M-9						9.5					0.6									
14M-10						3.3														
14M-11																				
14M-12																				
14M-13																				
14M-14																				
14M-15																				
14M-16																				
14M-17																				
14M-18																				
16M-1							2.0								21					
16M-2							1.0						T		37					
16M-3					T		T						T		19					
16M-4													T		31					
16M-5							T						1.1		65					
16M-6							1.2						1.0		40					
16M-7					T		29						T		4.0					
16M-8					T								T		27					
16M-9					35		100		T	T			5.0	2.1	74					
16M-10					37		80		T	3.2			7.4	3.6	56					
16M-11	4.6	10	52		124		330		45	550		31	170	53	41					
16M-12	66	120	18		555	1.5	1 450		210	975		13	164	150	6.0		T	T		
16M-13	7.0	1 510	8.0		1242	3.5	3 125		70	63			11	40						
16M-14	36	320	2.0		87		620		14	700		120	108	33	11	49	66	25	2.5	B
16M-15	200	1 950	160		264	5.0	3 850		140	1660		380	580	200	260	380	265	315	11	B
16M-16	33	330	T		6.0		640		13	140		70	100	38	190	58	48	15	0.8	B
17M-1							2.4						0.6							
17M-2						T	3.6						0.9							
17M-3							3.0			0.5			0.7							
17M-4							1.9			1.1			0.5							
17M-5							2.6			1.9			0.4							
17M-6							3.0			0.9										
17M-7							2.6			2.3			0.6							
17M-8							4.0			3.6			1.0	0.5						
17M-9				1.6		1.0	1.8			18			T							
17M-10			6.0	250		2.1	3 610			35			2.3							
17M-11			4.7	160		2.1	8.7			48				T						
17M-12				24			63		2.2	5.9										
17M-13				2.7		2.7	2 360		225	14			4.8	325						
17M-14	71			12		1.4	3 270		245	31			150	275	16					
17M-15	150			2.0		T	2 990		237	28		410	175	290	20					
17M-16	160						190		8.0	T		T	T							
17M-17							2 770		130	10		47	6.0							
17M-18				34																
18M-3					13						T		T							
18M-9					T			1.8			6.1	T	6.9	73						
18M-12					29			5.0	5.6											
18M-16					T			1.7												
18M-19												T	8.7	T						
19M-7						207	455		110			160	587	249			6.2	14	4.2	
19M-16						276	109		12	T		267	219	69			10	17	3.2	
19M-17																				

Table F-3. Continued

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CICL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEV	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
20M-1							7.8					4.8	5.8	1.3						
20M-2							25		0.6	T		15	18	7.2		1.2		0.5	1.0	
20M-11					38	T	180			T		5.7	7.0	5.0				T	0.6	
20M-12					88	22	2 080		8.0	14		3.3	4.1	4.3		0.4			T	
20M-13		1 170			83	27	1 630		152	219		16	40	17	2.3		6.0	T	T	
20M-14				1.4	T	213	110		21	15		58	71	425	15		6.5	6.8	7.8	
20M-15			T		T	87	74		4.8	37		270	414	200	48	0.8	6.5	22	38	
20M-16						T	24		1.7	8.6		60	73	45	8.0	7.6	40	3.8	0.8	
20M-17						T	5.0			0.6		T	5.0	6.5			T		T	
21M-1					T	2.5					67									
21M-2			5.2		96	18					220									
21M-3			2.9		12	82					805									
21M-4			9.0		690	3.9					280									
21M-5			13		845	8.1					423									
21M-6			7.6		30	6.6					460									
21M-7											9.9									
21M-8			4.0			3.3					42									
21M-9			9.0			T					42									
21M-10											T									
21M-11																				
21M-12																				
21M-13																				
21M-14																				
21M-15																				
21M-16																				
21M-17																				
21M-18																				
23M-4																				
23M-8							T		T	331										
23M-16			24	21							43									
33M-4			33		321	14		394			57									
33M-7			62		1023	11		822			12									
33M-15			2.0		61			66			26									
34M-2			4.9		380	23		460			104									
34M-3			29		2080	39		870			59									
34M-4			12		2460	22		10 530			71									
35M-5			16		2230	21		145			100									
34M-6			16		77	27					53									
34M-7			3.1		4.3	15		1.4			0.7									
34M-8					2.7	5.3		3.7			3.1									
34M-9						4.6		2.0												
34M-10								1.3												
34M-11						5.5		1.4			6.4									
34M-12						2.5		2.0												
34M-13					T	T		T												
34M-14					2.0			2.0												
34M-15					2.0			1.0												
34M-16					2.0			0.7												
34M-17																				
37M-1				2			59		T	4.3		28	24	71	8.4	T	1.6			
37M-2	T			3.9			83		2.3	5.0		30	30	94	9.7	2.6	2.0			
37M-3	6.7		T	9.1		2.2	220		9.0	7.0		31	31	115	8.2	2.5	1.9			
37M-4	5.0		T	31		2.2	665		20	8.8		31	34	128		T	2.3	T		
37M-5	T	490	2.9	375		8.3	3 430		150	36		22	24	1 140	15	T	3.5	T		
37M-6	370		7.5	1.0		12	3 491		220	75		33	26	1 540			6.0			
37M-7	620	2 030	7.3	730		14	7 940		450	55		31	27	2 950	127	2.0	13	6.0		
37M-8	10 400	9 370	23	1310			53 200		2000	185		48	32	10 300	740	21	13	T		A
37M-9	4 070	13 700	99	1980		29	33 900		1830	340		42	35	14 700	310	12	7.5			A
37M-10	180	890	7.6	190			1 875		105	115			13	766			T	6.0		A
37M-11	1 145	1 210	140	1100		14	10 040		620	1190		45	250	4 120	140	6.4	8.3	T		
37M-12	32	905	33	1230		12	2 240		490	88		19	83	2 890	63	2.6	16	T		
37M-13	232	630	5.0	1195		9.3	3 270		380	940		35	460	2 860	75	4.1	7.4	1.5		
37M-14	15	400	74	1020		6.2	1 570		175	615		328	202	3 197		3.6	14	0.8		
37M-15	10	1 120		33		3.2	135		24	10		6.7	134	646		1.0				

Table E-3. Continued

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEV	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
37M-16	69	1 250	45	30		3.8	4 540		140	68		95	740	164	1500	175	44	8.9	26	B
37M-17	60	760	4.9	26		3.3	4 750		108	77		600	825	90	2585	91	36	6.7		B
37M-18	43	500	6.6	24		3.0	4 150		73	79		920	760	63	2840	208	38	4.9		B
37M-19	35	520	5.0	65		3.5	3 325		110	55		1075	610	45	2475	140	18	2.1		B
38M-1					65			1 880		2.0			2.0							
38M-2			T		5.4		125	24					1.7							
38M-3					8.4		155	45	4.6				4.3							
38M-4					46			915	30	4.2			3.1							
38M-5					60			1 790	39	6.0			3.2							
38M-6					12		510	5.7	1.6				2.4							
38M-7					36			1 040	8.4	4.5			2.2							
38M-8					12		510	5.7	1.6				2.4							
38M-9					19		690	9.1	2.2				2.4							
38M-10					45			685	8.1	2.3			2.6	3.9						
38M-11					34	T		470	14	4.4			9.1	45						
38M-12					49	T		880	15	10			56	160						
38M-13			17		1190	8.1		940	16	47			69	220						
38M-14			3.5		270	3.4		1 380	24	16			18	63						
38M-15			5.4		433	5.7		1 530	22	17			13	46						
38M-16			7.0		622	6.9		1 240	16	23			7.9	22						
38M-17			4.2		446	5.6		1 180	18	22			8.0	22						
38M-18					65			440	14	3.7			4.6	13						
39M-1						2.1					2.5									
39M-2						5.9					47									
39M-3						9.8					7.8									
39M-5						0.7														
39M-6																				
39M-7																				
39M-8																				
39M-9																				
39M-10											T									
39M-11								2.2												
39M-12								2.9												
39M-13								2.9												
39M-14								2.5			T									
39M-15																				
39M-16								3.1												
39M-17								5.4												
39M-18								T												
39M-19																				
40M-8																				
40M-13																				
40M-18																	79			
47M-1						5.8											53			
47M-2											9.0									
47M-3						38														
47M-4						15														
47M-5						3.8											19			
47M-6						8.5														
47M-7																				
47M-8																				
47M-9																				
47M-10																	21			
47M-11																				
47M-12																				
47M-13																				
47M-14																				
47M-15																	40			
47M-16																				
47M-17																				
47M-18											6.6									
52M-4				3.1	119	5.2		208			1.6									
52M-14				T	54			275			8.2									
52M-17				2.2	98	T		333												

Table F-3. Continued

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEY	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
54M-1					6.8	T		T			3.0						40			
54M-2					14			19			6.4									
54M-3					T			5.9			5.0									
54M-4					4.0	5.2		5.4			23									
54M-5			8.4		37	6.6		665			22						37			
54M-6			12		260	3.2		655			13									
54M-7			12		280	5.2		635			26									
54M-8			16		470	4.0		1 110			23									
54M-9			11		550	4.5		1 370			25									
54M-10			2.7		310	T		1 055			7.5									
54M-11					80			525									135			
54M-12			T		42			270												
54M-13					T			290			T									
54M-14			2.1					210			4.4									
54M-15			T		57			85			3.0									
54M-16			T		50			240			T									
54M-17					2.5			155			T						82			
54M-18			T		6.5			310			4.0									
54M-19								T												
56M-4			T	150		52	848		93	56		52	41	253			126			
56M-7			T	98		T	212		21	52			29	119			7.9			
68M-1																				
68M-3					7.6															
68M-5					11															
68M-13					49															
68M-18					23															
68M-21																				

Table E-4. May-November 1983 Volatile Organic Analysis (µg/L)

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEV	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
14P-1																				
14P-3																				
32P-1																				
32P-2																				
32P-3						12														
32P-4						68														
32P-5																				
32P-6							T			T										
32P-7				T			95		22	52							T			
35P-1	46	890	3.5	325		T	4 450		325	13		1.0		1 220	1100		16			
37P-1																	16			
45P-1																				
45P-2																				
45P-3																				
48P-1																				
48P-2																				
48P-3																		T		
60P-1	20		12	250			1 000		150	700		17	130	60	20					
36W																	4.0			
44W																	4.0			
5M-1																	3.0			
5M-2																	2.1			
5M-3																	1.0			
5M-5																	1.2			
5M-6																	1.5			
5M-7																	10			
5M-8																	T			
5M-12																	0.8			
5M-14																	T			
5M-16																	T			
5M-17																	T			
5M-18																	T			
5M-19																	T			
12M-8																				
12M-10																				
12M-12																				
12M-13																				
12M-14																				
12M-15													T		35	T				
16M-1													6.0		31					
16M-2													T		18					
16M-3													T		43	T				
16M-4													T		35					
16M-5													T		30					
16M-6							22													
16M-7					T		51					12			11					
16M-8					37		71		T	T		14		T	12					
16M-9					34		52		3.0	T		60	9.0	T	12					
16M-10					174		410		170	700		35	180	48	16					
16M-11	T	10	11		211		490		130	740		20	140	79						
16M-12	9.5	120	9.0		925		1 540		82	74			13	44						
16M-13	T	1 510	5.4		211	T	980		63	1650		290	230	45	7.0	300	68	37	7.5	
16M-14	68	320	22		158	T	950		49	2195		110	160	58	35	140	15	18	1.6	
16M-15	48	1 950	26		167	T	1 160		56	2215		120	210	80	13	110	29	27	4.0	
16M-16	56	330	34								400									
21M-3			3.5		16	11														

Table E-4. Continued

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEY	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
37M-1	6.4						90		2.0	9.5			24	135		T	0.8			
37M-2	T			4.0			36		T	3.0			11	56		0.9	T			
37M-3				6.5			75		T	4.4		22	11	55						
37M-4	T	T		140			570		46	10		14		80						
37M-5																				
37M-6	7.0	1 300	10	400		13	2 900		290	36		66	7.0	3 770	1890		2.0	5.0		
37M-7	830	2 100	16	680			2 200		550	47		104	3.0	6 200	2500		4.0			
37M-8	9200	10 000	48	2350			41 700		1950	150		220	27	12 500	4870	2200	T			A
37M-9	4650	9 900	140	3090		T	38 500		2080	660		110	38	18 000	8400	1800				A
37M-10	2260	3 100	95	2300			22 600		1250	1860		84	295	12 100	3470	12	6.6	0.5		
37M-11	730	1 000	70	1540			9 000		620	3400		55	580	4 890	1100		8.0	T		
37M-12	370	530	65	2080			3 300		380	890		23	160	3 900	1190	4.0	160	T		
37M-13	100	450	68	2100		16	1 400		410	2500		60	2480	3 430	970	T	12	4.6		
37M-14	10	290	50	1370			740		200	1000		27	485	3 100		T	8.3			
37M-15	5.1	1 800	2.9	120		16	560		120	40		T	22	620		T	2.0			
37M-16	19	73	3.8	16		T	3 000		150	110			550	48	720	145	20		T	
37M-17	33	140	29	29			3 260		100	130			540	30	950	110	1.5	T		
37M-18							2 940		85	140		T	450	12	45	105	T			
37M-19	9.0	1 816	T	17													0.8			
52M-8			1.8		90			625		3.5										
52M-9			2.3		73			660												
52M-10					35			615												
53M-5			11		260			660		9.0										
53M-6			5.7		195			590		6.5										
53M-7			17		180			1030		7.5										
54M-5			13		200			620		12										
54M-6			9.0		175	T		695		16										
54M-7			22		185			495												
54M-8			6.3		160	T		500		8.0										
54M-9			4.3		165	3.7		970		8.0										
54M-10			4.5		198	T		951		8.0										
67M-1				4.0			16		5.0	8.0			74	53			11			
67M-3		25		21			42		19	11			74	200			12	T		
67M-4		25		16			44		15	11			79	145			13			
67M-5		20		21			49		15	5.0			48	160			7.5			
67M-6		120		120			181		39	15			86	340		T	13	T		
67M-8	13	600		100			475		60	35		T	93	98			19	5.3		
67M-9		2 750	19	460			832		140	410		11	1820	1 650			260	T		
67M-10	18	2 240	7.5	345			1 167		165	200		45	620	560	T	T	90	15		
67M-12	170	3 600	5.5	250			2 113		230	230		13	300	405	T	T	18	7.5		
67M-14	22	880	4.0	125			695		110	110		20	285	335		T	6.5	6.5		
67M-15	T	25	4.0	37			300		80	85		38	250	195		T	1.6	7.0		
67M-17	2.0	190	1.0	45			290		48	56		25	126	140			4.8	2.8		
67M-18	3.5	540	T	60			400		55	70		36	140	130			4.0	2.0		
67M-19	8.0	550	1.6	48			322		52	60		9.0	120	160			4.7	2.5		
67M-20		15		12			38		27	39		75	84	44			1.7	2.0		
67M-22				5.7			37		11	51		17	6.9	6.1			T			
67M-23				1.2			28		8.5	20		19	13	4.9			T			
67M-24				5.0			199		23	25		21	56	19			T	T		
68M-1																				
68M-2																				
68M-3																				
68M-4																				
68M-5					33															
68M-6					7.7															
68M-7																				
68M-9																				
68M-10																				
68M-11					1.0															
68M-12					140															
68M-17																				

Table E-4. Continued

Site No.	CH2CL2	ACET	11DCEE	11DCEA	THF	T12DE	CHCL3	DEE	12DCEA	111TCEA	14DNE	CCL4	TCEY	BENZ	CHBR3	TECE	MEBEN	CLBEN	ETBEN	Notes
68M-18																				
68M-19					17															
68M-20					23															
68M-21					T															
68M-22																				
68M-23											T									
83M-1						2.6					12									
83M-3						1.2					7.6									
83M-5						T					1.5									
83M-7											9.0									
83M-9											T									
83M-11																				
84M-0																				
84M-2																				
84M-4																				
84M-5																				
84M-6																				
84M-7																				
84M-19																				
85M-0																				
85M-1																				
85M-1																				
85M-2																				
85M-4																				
85M-6																				
85M-15																				